

Ecosystem Recovery Planning for Listed Salmon: An Integrated Assessment Approach for Salmon Habitat

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EXECUTIVE SUMMARY

The Endangered Species Act (ESA) requires that a recovery plan be developed for all listed threatened and endangered species. For Pacific Salmon, this includes 26 evolutionarily significant units (ESUs) of five species of salmon and of steelhead trout distributed among nine geographical areas along the West Coast. Several factors associated with harvest, hatchery, hydropower, and habitat have been identified as important to understanding the decline in salmon and should be addressed in a successful recovery plan.

Previous documents have provided guidance for recovery planning (McElhany et al. 2000, NMFS 1992, 2000). However, they don't address specific guidance on how to implement the habitat portion of a recovery plan. This technical memorandum supplements prior guidance documents with information specific to habitat recovery planning. It is not intended that existing habitat recovery planning efforts (e.g., at the local watershed level) should be abandoned in favor of methods discussed here. Rather these methods should help clarify the specific purposes and methods of assessment within their existing approaches. Audiences that may benefit from this document include Technical Recovery Teams (TRTs) assigned to each of the geographical planning areas, local watershed planning groups, and National Marine Fisheries Service (NOAA Fisheries) personnel. Although not addressed in this document, information provided by following the guidelines in this document may be beneficial in other ways, such as increased compliance with environmental regulations (including other portions of the ESA).

An Assessment Approach for Habitat Recovery Planning

Our conceptual approach to habitat recovery planning is based on principles of watershed and ecosystem function. Salmon are adapted to local environmental conditions (including

associated temporal and spatial variability), and those conditions vary in space and time due to landscape processes and land use. Because landscape processes (e.g., sediment supply, wood recruitment to streams) create and sustain habitats over time, an approach to habitat recovery that focuses on restoring or preserving ecosystem processes should provide good quality salmon habitat over the long term. This general approach applies to all ecoregions, although the relative importance of various landscape processes differs by ecoregion.

Recovery planning has been separated into two phases: Phase I planning identifies recovery goals and Phase II planning identifies recovery actions. For Phase I planning, a suite of habitat analyses can help identify or clarify certain recovery goals (e.g., abundance goals) for ESUs or populations. The Phase I assessments can also identify where large habitat losses have occurred and may help identify which habitats limit recovery of populations. In Phase II planning, watershed process assessments are the basis for identifying causes of habitat loss or degradation, as well as ecosystem recovery actions. Results from both assessments can be used to prioritize restoration actions, and new information gained from future assessments and management experiments should be used to update the recovery plan in the future.

Habitat Analyses for Phase I Recovery Planning: Setting Recovery Goals

The first phase of habitat recovery planning addresses two main types of questions:

1. How might habitat changes have altered the abundance, survival, population growth rate, spatial structure and life history diversity of ESUs or individual populations?
2. What scenarios of habitat characteristics would support a viable ESU?

These questions can be addressed by assessments and analyses conducted at several levels of resolution. First, ESU-wide analyses can provide an overall understanding of broad-scale

patterns of land use and habitat conditions, and relate these to the recovery goals (salmon abundance, survival, population growth rate, life history diversity, and spatial structure). Second, watershed-level analyses can elucidate such patterns specific to each watershed. Results of these analyses can be used to set biological delisting criteria for each of the salmon ESUs and their constituent populations.

The ESU-level analyses are meant to provide information about a broad geographic area in a relatively short period of time using existing data. Consistent methodologies applied across and entire ESU enables comparisons of results among populations or watersheds. Correlation analyses using existing geo-spatial data can identify relationships among natural landscape attributes, land uses, and salmon populations (e.g., Salmonid Watershed Analysis Model). Comparisons between historic and current habitat conditions can help assess potential productivities and capacities of salmon populations, identify where large habitat losses have occurred, and help identify which habitat losses might have large affects on ESU viability.

At the watershed level, similar analyses can be conducted to ascertain relationships among landscape attributes, land uses, and salmon viability. Existing tools include simplified limiting factors models, the Ecosystem Diagnosis and Treatment (EDT) model, and dynamic life cycle models. All three approaches compare historic and current habitat conditions, but differ in their data requirements and representation of the salmon life cycle. EDT is a complex model that greatly relies on expert opinion for input data, whereas the other approaches are simpler models based primarily on measured data. The simplified limiting factors model has the least complete representation of the salmon life cycle, allowing only life-stage capacities to change. EDT allows both life-stage capacities and survivals to change. The dynamic life cycle model allows both life-stage capacities and survivals to change, and can examine population growth rates over

time in response to habitat actions. These assessments elucidate patterns of habitat alteration and highlight areas where such change may have most greatly affected salmon viability. Results can be used to identify criteria required to sustain viable populations, and identify critical uncertainties in our predictions of how populations will respond to habitat restoration.

Habitat Analyses for Phase II Recovery Planning: Identifying Ecosystem Restoration Actions

Phase II assessments are primarily intended to identify causes of habitat loss or degradation and to identify necessary ecosystem restoration actions. Specific inventories and assessments to identify altered ecosystem processes can be grouped into distributed processes (i.e., widespread non-point such as sediment supply inventories), reach-level processes (e.g., floodplain and riparian characterization), and other ecosystem functions not easily described by rates or levels (e.g., barrier and flow-diversion inventories). Assessments that identify impaired biological integrity (e.g., Benthic Index of Biological Integrity or B-IBI, multivariate model analyses) can identify locations where habitat degradation may be altering biological communities and which ecosystem processes have been disrupted. All of these analyses aim to identify the natural landscape processes active in a watershed, the effects of land use on natural processes, and the causal relationships between land use and habitat conditions. Specific results include identification of locations where stream segments, reaches, or subwatersheds are most impaired, and causes of impairment. From these assessments, a list of recovery actions can be prepared for each watershed in an entire ESU.

Prioritizing Potential Restoration Actions within Watersheds

Many factors influence the prioritization of restoration actions in recovery planning, and there are many philosophical approaches to recovery of ecosystems and listed species. When little is known about habitats that limit recovery of listed ESUs or the causes of habitat degradation, an interim prioritization approach based on effectiveness of different types of actions can be used. This hierarchical strategy gives priority to actions that have a high probability of success, low variability among projects (i.e., consistency of results), a relatively quick response time, and a long duration of results. If more is known about habitat limitations and causes of habitat loss, Phase I and Phase II information can be combined to prioritize restoration actions necessary for more efficient recovery of single species. Cost and time required to implement actions as well as immediate management needs may also be considered during prioritization. Where more than one species is of concern or where habitat recovery goals are more broadly defined, alternative strategies that consider multiple species, protection of refugia, and other factors may be used.

Uncertainty in Habitat Recovery Planning

Acknowledging, describing, and estimating uncertainty associated with assessments and analyses can increase the effectiveness of recovery planning. This process elucidates the full range of possible outcomes and the probability of seeing each of these outcomes. Knowing where uncertainties exist allows managers to develop plans with acceptable risk (i.e., plans where the benefits of an action outweigh its costs).

By recognizing where uncertainty exists, areas that need further clarification (e.g., more data, additional expert opinion, better model performance) can be identified. To do this,

estimates of the magnitude of uncertainty in each of five types of uncertainties must be generated: 1) prediction, 2) parameter, 3) model, 4) measurement, and 5) natural stochastic variation. Identification of which types of uncertainty are likely to have the largest effect on predictions can suggest areas where improvements in information will be most beneficial.

Often, decisions need to be made before adequate data are available. Provided that uncertainties are identified, several established methods can be used to make decisions based on the best available information. These methods aid in prioritization of actions, and are preferred over methods that rely on guesswork, biased data, or data collected at inappropriate scales. Final outcomes chosen must be robust to each of the types of uncertainty identified. These decision strategies should assist in creating sound plans in the interim, and can be re-evaluated as new information is obtained.

Appendices

This technical memorandum includes three appendices. Appendix A, Issues of Scale in Habitat Recovery Planning, examines the concept of scale in recovery planning with particular emphasis on analyses to help set recovery goals. Appendix B, Estimating Chinook Salmon Spawner Capacity of the Stillaguamish River, is an example analysis for Phase I recovery planning. Appendix C, Restoration of Habitat-forming Processes: An Applied Restoration Strategy for the Skagit River, is an example analysis for Phase II recovery planning.

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INTRODUCTION

One of the main purposes of the Endangered Species Act (ESA) of 1973 is “to provide a means whereby the ecosystems upon which endangered species and threatened species depend may be conserved.” The ESA consequently requires the development and implementation of recovery plans in order to affect the conservation of listed species, and details that recovery plans must include:

- 1) a description of such site-specific actions as may be necessary to achieve the plan’s goal for conservation and survival of the species;
- 2) objective, measurable criteria which, when met, would result in a determination that the species be removed from the list; and
- 3) estimates of the time and cost required to carry out those measures needed to achieve the plan’s goal and to take the intermediate steps toward that goal.

For ESA-listed salmon in the western United States, this requirement is no small task, as salmon habitat is ubiquitous and the actions required to protect or restore the ecosystems on which salmon depend are in conflict with many land uses in the region.

The ESA provides little guidance concerning the content of recovery plans for individual species. Therefore, the National Marine Fisheries Service (NOAA Fisheries) provides additional scientific guidance on setting recovery goals for evolutionarily significant units (ESUs) of salmon and the populations within them (McElhany et al. 2000), based on the concept of viable salmonid populations (VSPs). An ESU, equivalent to a “distinct population segment” under the ESA, is “a population or group of populations that are 1) substantially reproductively isolated from other populations, and 2) contribute substantially to the ecological or genetic diversity of the biological species” (Myers et al. 1998). For each ESU, recovery goals generally are

concerned with identifying how many and which independent populations are necessary for ESU viability (McElhany et al. 2000). McElhany et al. (2000) identify four categories of recovery goals that must be met for each population within an ESU of listed salmon: population abundance (size), population growth rate (and related parameters), spatial structure (within a population or group of populations), and diversity (i.e., the distribution of traits within and among populations). However, the VSP guidance did not address how to identify specific recovery actions for harvest, hydropower, hatcheries, or habitat that were necessary to achieve ESU or population viability.

In addition to the VSP guidance, NOAA Fisheries provides guidance to the Technical Recovery Teams (TRTs), which are tasked with developing the technical aspects of a recovery plan for each ESU of listed salmon (NMFS 2000, referred to as the TRT Guidance Document). This guidance document identifies two phases of recovery planning: Phase I identifies the recovery goals (i.e., criteria that must be met for delisting), and Phase II identifies restoration actions that will be necessary for recovery. However, there is considerable overlap in habitat analyses used for Phase I and Phase II planning. The habitat elements of the TRT work program are mainly included in Phase II planning, and are identified as:

- 1) describe fish and habitat relationships,
- 2) identify factors causing decline and limiting factors, and
- 3) identify actions for recovery.

The TRT Guidance Document goes on to indicate that characterizing habitat/fish relationships includes assessing the spatial distribution of fish abundance for each population in the ESU, associating fish abundance with habitat characteristics, and identifying human factors that have the greatest impact on key freshwater and marine habitat. However, it does not specify

appropriate spatial scales or resolution levels of data analyses. Moreover, it does not clearly elucidate the questions that such analyses are intended to answer, especially as they relate the population goals for diversity and spatial structure. The TRT Guidance Document also stops short of specific questions for identifying limiting factors and identifying habitat recovery actions. As neither the VSP or TRT Guidance Documents addressed how to identify specific habitat actions that would support salmon recovery, the Watershed Program at the Northwest Fisheries Science Center was asked to develop guidance for the habitat restoration elements of salmon recovery plans. In response to that request, this technical memorandum describes an approach to developing the habitat elements of a recovery plan for salmon listed under the ESA.

Purpose and Scope

The purpose of this document is to help both TRTs and local watershed groups list specific recovery planning questions, assemble appropriate methods for answering those questions, and utilize that assessment information to identify and prioritize ecosystem recovery actions. We focus on analyses that can help identify the restoration actions necessary to recover ecosystems that support salmon, as well as help set population recovery goals. However, there are many other aspects of recovery planning that we do not address here, including the Columbia River hydropower system, exotic species impacts, or harvest and hatchery practices. We also do not specifically address certain regulatory statutes that may be considered programmatic elements of a recovery plan. Regulations such as water quality standards, forest practices rules, the Northwest Forest Plan, and local growth management ordinances should serve as ecosystem protection actions at a minimum, and may serve as passive recovery actions in the best case (e.g., where substantial riparian buffers allow natural recovery riparian processes and functions). A

companion document provides a context for the implementation of data collection and analysis methods such as those described here (Fullerton and Golde in prep.).

Here we provide guidance for two main audiences, TRTs and local watershed groups (e.g., watershed councils, lead entities, etc.) that identify and conduct restoration actions. In general, we expect TRTs to focus largely on analyses for Phase I recovery planning (with some overlap into Phase II) and local watershed groups to focus mainly on analyses for Phase II recovery planning. We recognize that many existing assessment methodologies already incorporate aspects of the guidance that we provide here (e.g., Moore 1997, Quigley and Arbelbide 1997, SWC 1998, OWEB 1999a, JNRC 2001), and we support existing approaches that favor restoration of ecosystem processes and functions. We do not intend that existing methods be abandoned in favor of a redesigned assessment, but we believe that this guidance will be useful to existing TRTs and local watershed groups in clarifying the specific purposes and methods of assessment within their existing approaches. In addition, there are many TRTs yet to be formed and many local watershed groups that have not yet formulated an approach and methodology for recovery planning.

Overview

This document provides guidance for choosing and conducting analyses to assist in both Phase I and Phase II recovery planning for listed salmon. In the next section, “An Assessment Approach for Habitat Recovery Planning,” we begin with a conceptual framework for understanding relationships among land uses, watershed functions, habitat conditions, and biota. The conceptual framework relies on principles of watershed and ecosystem management, and organizes the habitat-related questions that each recovery plan should attempt to answer. These questions first address how habitat changes might have affected abundance, survival, population

growth rate, spatial structure, and diversity of salmon populations within an ESU (questions relevant to Phase I recovery planning or setting recovery goals). A second group of questions addresses causes of habitat change, which provides the basis for identifying restoration actions that are necessary to recover the ecosystem upon which salmon depend (Phase II recovery planning).

After listing the important questions, we provide a brief overview of methodologies that are appropriate for answering each question. In the “Analyses for Phase I Recovery Planning: Setting Recovery Goals” section, we discuss analyses that help set recovery goals and create a broad understanding of habitat issues affecting salmon populations across an ESU. We then describe more detailed assessments to be conducted within individual watersheds that identify causes of habitat loss or degradation (the “Analyses for Phase II Recovery Planning: Identifying Ecosystem Restoration Actions” section), and how Phase I and Phase II information can be used together to help prioritize restoration actions (the “Prioritizing Potential Restoration Actions Within Watersheds” section). Finally, in the “Uncertainty in Salmon Habitat Recovery Planning” section, we discuss how uncertainty may affect planning decisions and provide guidance and examples for identifying and quantifying types of uncertainty. This guidance document also includes three appendices that discuss 1) issues of scale in habitat recovery planning, 2) an example analysis for Phase I recovery planning, and 3) an example analysis for Phase II recovery planning.

Important Notes on Terminology

Because salmon recovery planning draws on many scientific disciplines there is considerable variation in use of terms (e.g., the term productivity has different meanings, depending on the discipline in which it is used). To help minimize confusion surrounding

specific terms, we have included a glossary to clarify their meanings as we use them here. Five terms commonly used in salmon recovery planning that we wish to explain now to help avoid misinterpretation of our guidance are habitat, ecosystem, recovery, restoration, and productivity.

The first two terms, habitat and ecosystem, are often used interchangeably, which creates some confusion about their meaning. For the purposes of this document, the term habitat refers to the aquatic environment that fish experience and not those landscape processes or attributes outside streams that alter habitat conditions. In general use, the term ecosystem refers to the dynamic and holistic system of all the living and dead organisms in an area and the physical and climatic features that are interrelated in the transfer of energy and material. In this document, it is the aquatic environment and biota, physical and biological processes active in that environment, and the landscape processes and land uses that form and sustain the aquatic environment and biota. In general, recovery planning will aim to restore habitat attributes that support salmon by restoring ecosystem processes that form and sustain those habitats.

The third term, recovery, in the context of listed populations, means attaining specified goals for viable populations and ESUs (abundance, population growth rate, spatial structure, and diversity). For watershed processes and habitats, recovery means returning from a disturbed state to some prior condition, not necessarily pristine.

Restoration, the fourth term, in its strictest definition, is returning a site to some predisturbance condition (Gore 1985, NRC 1996). Some practitioners call this full restoration. It is generally more holistic or systemic than habitat creation, reclamation, rehabilitation, or enhancement, and not accomplished through manipulation of individual ecosystem or watershed elements. In contrast, habitat enhancement is the improvement of habitat from its existing or previous condition. It does not necessarily seek to restore conditions to some predisturbed state

or seek to restore disrupted watershed or ecosystem processes and functions such as delivery of water, wood (organic material) and sediment. Some practitioners call this partial restoration. Here we use the term restoration generically to mean both restoration and enhancement, but we distinguish between those activities that restore watershed or ecosystem processes and those that enhance habitat.

The fifth problematic term is productivity. In salmon management and research it has four meanings: population growth rate (e.g., McElhaney et al. 2000), number of adult returns per spawner, (e.g., Moussalli and Hilborn 1986), stage to stage survival rate at low population size or density independent survival (e.g., Moussalli and Hilborn 1986, Lestelle et al. 1996), and plant and algae biomass produced per unit area per year (e.g., Begon et al. 1986). To reduce the potential for misunderstanding, we have chosen not to use the term productivity and instead use more specific terms as appropriate.

AN ASSESSMENT APPROACH FOR HABITAT RECOVERY PLANNING

In this section we briefly describe an approach to understanding ecosystem functions and habitat change, as well as the scientific and practical reasons for choosing this approach. For the purposes of this document, the term habitat refers to the aquatic environment that fish experience, and not those landscape processes or attributes outside streams that alter habitat conditions. The ecosystem we discuss encompasses freshwater and delta habitats, as well as the landscape processes and land uses that form and sustain those habitats. Our approach is based on a simple conceptual framework for understanding relationships among ecosystem processes, land uses, habitat conditions, and biota. Using this conceptual framework, we organize a series of questions that must be answered in developing a recovery plan, identify the purpose of each assessment method, and illustrate the relationships among different assessments. We also suggest a sequence for the assessments and describe the importance of management experiments and monitoring in updating the recovery plan.

Restoring Ecosystems to Support Recovery of Listed Salmon

Over the past decade, many scientists have pointed out that the listing of salmon and other species as threatened or endangered is largely a result of trying to manage individual species and habitat characteristics rather than managing whole ecosystems (e.g., Doppelt et al. 1993, Frissell et al. 1997). Scientists and managers alike have recognized that restoration that carefully considers the watershed or ecosystem context is more likely to be successful at restoring individual or multiple species and preventing the demise of others (Nehlsen et al. 1991, Doppelt et al. 1993, FEMAT 1993, Lichatowich et al. 1995, Reeves et al. 1995, Beechie et al.

1996, Moore 1997). These conclusions suggest that habitat recovery planning will require assessments of disruptions to ecosystem functions and biological integrity, which have reduced the productive capacity of Pacific Northwest river systems and are partly responsible for the declines in salmon abundance. The goal of such assessments is to identify alterations of key processes that affect stream habitats, and specify the management actions required to restore those processes that sustain aquatic habitats and support biological integrity (e.g., FEMAT 1993, Moore 1997, Quigley and Arbelbide 1997, Beechie and Bolton 1999). In this approach, restoring specific salmon populations (or any other single organism) is subordinate to the goal of restoring the ecosystem that supports multiple salmon species. In addition, information on habitat changes or conditions that limit specific salmon populations can be useful for identifying actions that may have the greatest effect on salmon recovery (e.g., Reeves et al. 1991), or for helping to set population and ESU recovery goals. As long as all restoration actions are consistent with the overriding goal of restoring ecosystem processes and functions, habitats will be restored for multiple species, but in a sequence that favors one over the others.

For this report, the ecosystem approach to salmon recovery planning includes two main assessment elements: analysis of landscape and habitat factors to help set recovery goals, and analysis of disrupted ecosystem processes to identify watershed and aquatic habitat recovery actions. Each type of analysis relies on a conceptual framework describing general relationships among land uses, landscape characteristics, aquatic habitat, and biological responses (Figure 1). This framework illustrates that landscape processes and land uses alter aquatic habitats, which in turn alter aquatic communities or populations. Therefore, aquatic habitat conditions can be viewed as the link between landscapes and fish populations. Making these relationships explicit allows us to organize analyses of ecosystem processes and functions in a way that brings greater

clarity of purpose to each analysis, as well as a better understanding of how the results of each analysis are to be used in recovery planning.

Scientific Basis for an Ecosystem Approach

The scientific basis for this approach can be summarized in two important characteristics of salmon and their habitats (Beechie et al. 1996):

1. Salmonid stocks are adapted to local environmental conditions, including the dynamic nature of their environment (Miller and Brannon 1982, Healey 1991, Reeves et al. 1995).
2. Spatial and temporal variations in landscape processes create a dynamic mosaic of habitat conditions in a river network (e.g., Naiman et al. 1992).

These statements imply that salmonid species or stocks are adapted to spatially and temporally variable habitats (Beechie et al. 1996), and may further imply that such environmental variability is important to the long-term survival of stocks or races (Reeves et al. 1995). Perhaps most importantly, different salmon populations (even some located very close to each other) are adapted to the different spatial and temporal sequences of habitat conditions found in each watershed, which influences life history diversity across an ESU.

Because salmonids are adapted to spatially and temporally varied habitat conditions, it does not make sense to manage for the same conditions in all locations, or to expect conditions to remain constant in any single location. This has been recognized in scientific critiques of many management issues in the past decade, including “one-size-fits-all” habitat standards (Bisson et al. 1997), not managing for spatial or temporal variation in habitats (Reeves et al. 1995, Bisson et al. 1997), and addressing symptoms of a disrupted ecosystem rather than the causes (Frissell and Nawa 1992, Spence et al. 1996). Those approaches generally do not consider that local

populations are adapted to the natural potential habitat conditions within their range, and that those conditions vary in space and time. By contrast, identifying the root causes of degradation (i.e., impaired ecosystem processes and functions) focuses restoration on those processes that form and sustain habitats, which allows each part of the river network to express its natural potential habitat, and helps conserve and restore the natural spatial and temporal variation of habitats to which salmon are adapted.

We stress that identifying the root causes of ecosystem degradation is important for two main reasons. First, scientists and resource managers do not understand most of the linkages between landscapes, habitat, and salmon populations with any great certainty, and we cannot predict exactly how land uses alter habitat conditions or how those habitat changes alter salmon populations. In fact, it can be argued that we are not yet even aware of all the aspects of aquatic ecosystems that significantly affect salmon populations. This lack of knowledge has in the past led to significant habitat degradation. For example, the role of wood debris in habitat formation was poorly understood until the 1970s. Consequently, removal of wood from rivers for navigation over the past 150 years has resulted in dramatic alteration of river habitats (Sedell and Luchessa 1982, Collins et al. 2002), and as recently as the 1980s biologists recommended widespread wood removal to help adult salmon migrate upriver. While wood removal is far less common presently (but still occurs), the example serves to illustrate that we could have avoided significant habitat loss by choosing management actions that preserved riparian forest processes and natural wood functions in channels, even without understanding the value of wood in aquatic ecosystems.

Second, traditional restoration actions such as bank protection or spawning gravel placement attempt to build habitats that do not move in space or time, whereas natural habitats

are often created by movement of river channels, wood debris, and sediment. Therefore, many restoration actions fail to restore habitats because they do not recognize the integrated nature of physical and ecological processes in watersheds (Frissell and Nawa 1992, Beechie et al. 1996). This lack of knowledge leads to two main types of failure: 1) site-prescribed engineering solutions can be overwhelmed by altered watershed processes that are far removed from degraded habitats (e.g., increased sediment supply from upslope sources can bury engineered structures and pools), or 2) such measures can prevent habitat formation that would otherwise naturally occur (e.g., bank protection prevents formation of new off-channel habitats). Avoiding these types of project failure requires that we focus on restoring ecosystem processes and functions that form and sustain salmonid habitats, rather than on the habitats themselves.

Many organizations have recently adopted approaches to salmon habitat restoration that have a watershed or process-based approach (e.g., Moore 1997, SWC 1998, OWEB 1999a, JNRC 2001), which should help avoid some of the mistakes just described. However, many local groups continue to identify restoration projects in an opportunistic fashion, making difficult to assemble a broader understanding of habitat degradation and decline of listed species. Without this larger context, proposed projects are often disconnected from each other and fail to address the most important habitat losses. Continued development of holistic assessment approaches at the watershed level (e.g., SWC 1998, OWEB 1999a) should help resource managers to more effectively utilize funding and resources allocated for salmon recovery.

Practical Considerations

Systematic watershed assessments can address several parts of a salmon recovery plan. First, three types of watershed assessments address tasks listed in the TRT guidance document:

- 1) correlation analyses can help in setting population recovery goals (Phase I planning) and identify fish and habitat relationships (Phase II planning),
- 2) assessments of historical and current habitat abundance and quality can help identify recovery goals (Phase I planning) or factors causing decline and limiting factors (Phase II planning), and
- 3) assessments of altered habitat-forming processes help identify causes of habitat loss or degradation and actions for ecosystem and habitat recovery (Phase II planning).

In combination, these assessments provide a broad understanding of actions that are likely to improve population performance of listed salmon, and form the basis of both regional and site-specific plans for ecosystem restoration. Beyond the TRT guidance tasks, systematic watershed assessments provide the agencies a watershed-level understanding of habitat restoration needs, which they can use for evaluating habitat conservation plans, programmatic actions, and proposed habitat restoration projects (consultations).

These assessments can also help agencies address the Clean Water Act (CWA) by identifying causes of various water quality problems, especially those associated with non-point pollution sources. The assessments are consistent with more holistic management approaches such as watershed management (e.g., Swanson 1981), ecosystem management (e.g., Johnson et al. 1985), and managing for biodiversity or biological integrity (e.g., McNeely et al. 1990, Karr 1991), and can simultaneously help accomplish the habitat-related purposes of both the ESA, passed by Congress in 1972, and the CWA, passed by Congress in 1972. The stated purposes of the acts are to “provide a means whereby the ecosystems upon which endangered species and threatened species depend may be conserved” (ESA), and “restore and maintain the chemical, physical, and biological integrity of the nation’s waters” (CWA). The common thread is that

management of landscapes and ecosystems is a single approach that will produce sustainable clean water and support salmon recovery.

Several watershed assessment approaches recently adopted by a variety of salmon recovery groups in the Pacific Northwest (e.g., FEMAT 1993, WDNR 1995, SWC 1998, OWEB 1999a, JNRC 2001) focus on restoring ecosystems and watershed processes, and should help address the parallel goals of the ESA and CWA more efficiently. These watershed assessment processes support an ecosystem approach, or at least include certain process-based components of an ecosystem approach. While some of these assessments were not specifically designed to help develop Pacific salmon recovery plans, they provide data that are relevant to understanding disruptions to watershed or ecosystem processes. To the extent that these assessments answer specific questions relevant to habitat recovery planning, their results can be used within the context of the approach described in this report.

Key Assessments for Habitat Recovery Planning

Developing the habitat elements of a salmon recovery plan requires an understanding of how land uses have altered landscape processes that form and sustain salmon habitats, and how those habitat changes might have affected salmon populations. In this section we describe two groups of questions that must be answered to develop a habitat recovery plan (Table 1). The first set of questions is important in Phase I recovery planning, and concentrates on how habitats have changed since pre-settlement times and how those habitat changes have affected salmon and other biota. These questions motivate historical reconstructions of habitat types and abundance, as well as assessments of relationships between habitat and salmon population characteristics. The second group of questions focuses on identifying disruptions to ecosystem function and the types of habitat restoration necessary for ecosystem recovery. These questions are important in

Phase II recovery planning and motivate assessments that identify where the biological integrity of ecosystems has been degraded and where specific ecosystem processes or functions are disrupted.

For organizational purposes it is useful to diagram the relationships among habitat assessments that can be used in recovery planning (Figure 1). The Phase I assessments (those regarding changes in habitat and salmon populations) fall into two groups: 1) assessments that quantify habitat change and then use habitat-based models to estimate changes in fish populations (e.g., limiting factors analysis, life-cycle models, and Ecosystem Diagnosis and Treatment or EDT), and 2) correlation analyses that relate landscape and land use characteristics to fish population performance without directly quantifying changes to habitat (e.g., the Salmon Watershed Analysis Model or SWAM). It should be noted that neither of these assessments directly identifies causes of habitat degradation or specific restoration actions. However, these assessments have several important uses in setting the recovery goals for the ESU and each population within it (i.e., Phase I recovery planning). First, they provide habitat-based estimates of potential population size for comparison to estimates from population viability analyses (see McElhany et al. 2000 for background on use of population viability analyses in describing VSPs). Second, they provide insights into potential changes in life history diversity by identifying losses of important habitat types. And third, the ESU-wide correlation analyses can be used to help identify which populations are most constrained by habitat loss and therefore may be most difficult to recover.

The Phase II assessments (those regarding ecosystem functions and biological integrity) can be separated into two components: 1) screening assessments to that identify areas where ecosystem processes and functions are most impaired, and 2) specific field inventories to

diagnose causes of ecosystem impairment (see Appendix C, page XXX, for examples).

Assessments that correlate landscape and land use characteristics with population attributes (e.g., SWAM) can indicate which habitat changes are most likely responsible for declines in salmon populations, and therefore which broad categories of restoration actions are most likely to result in increased salmon populations. Direct assessments of ecosystem processes that form salmon habitats (e.g., barrier inventories, riparian condition inventories) identify causes of degradation, as well as restoration actions that are required to recover ecosystem functions and biological integrity.

It is important to note that, while there are relatively few differences in Phase I assessment procedures, Phase II assessment procedures can vary substantially across the Pacific Northwest. Pacific Northwest environments have been classified as a nested set of ecoregions with varying levels of detail (CEC 1997) (Figure 2). At the coarsest levels (Levels I and II), these ecoregions denote three main areas within which climate, lithology, topography, and ecosystems are generally similar: marine northwestern coastal forests, drier northwestern forested mountains, and semi-arid to arid western deserts (CEC 1997, USEPA 2000). Basic differences in watershed process assessments among ecoregions are shown in Table 2 (see also OWEB 1999a). In general, the same categories of assessments must be conducted regardless of ecoregion (e.g., sediment supply, riparian functions, isolated habitats), but the specific processes or mechanisms addressed may vary from one ecoregion to another. For example, sediment supply to the stream network should be evaluated in any watershed, but certain processes of sediment supply may be emphasized depending on location. Sediment supply is dominated by landsliding in most watersheds of the coast range and Cascade Mountains (e.g., Sidle et al. 1985), so understanding land use effects on landslide rates and sediment volumes is critical to

necessary restoration actions such as road decommissioning or reconstruction. By contrast, sediment supply in dry rangelands of the Columbia Plateau is more a function of surface erosion and gullying (e.g., Kaiser 1967, Peacock 1994), especially where soils are bare for some portion of the year due to agricultural practices. In these areas, assessing sediment impacts will focus more on changes in surface erosion rates and volumes in order to identify where modification of agricultural practices may reduce sediment supplies. These and other analyses of watershed processes will be described in more detail in the “Habitat Analyses for Phase II Recovery Planning: Identifying Ecosystem Restoration Actions” section of this document, page XXX, and the “Prioritizing Potential Restoration Actions Within Watersheds” section, page XXX.

Sequencing the Assessments

The typical sequence of assessments (illustrated in Figure 3) is:

- 1) Phase I: identify recovery goals for populations and ESUs, including goals for abundance, population growth rate (and related parameters), spatial structure, and diversity;
- 2) Phase II: evaluate ecosystem processes and functions to identify the suite of possible habitat recovery actions;
- 3) Prioritize recovery actions using Phase I information or other prioritization schemes (see the “Prioritizing Potential Restoration Actions Within Watersheds” section).

Phase I assessments indicate where the greatest habitat losses have been, what types of habitat losses have occurred, and which habitats are likely having the greatest affect on individual salmon populations. They are not ecosystem assessments, and focus on individual species in order to establish to the four categories of recovery goals. Phase I assessments are typically conducted at two levels of resolution: 1) ESU-wide correlations among landscape attributes and

fish population characteristics (e.g., abundance, life history patterns) using low-resolution geospatial data, and 2) more detailed watershed-level assessments of habitat availability for different life history stages. The former provides information that is useful in setting goals for recovery the entire ESU, whereas the latter is focused on setting more specific recovery goals for individual populations. Typically an ESU-wide assessment will be completed first because information needs are limited mainly to existing data. These assessments can indicate where the most important habitat losses have been, and may provide some insight as to which habitats may be limiting individual populations. Referring to Figure 1, these assessments will relate landscape attributes and land use practices directly to fish abundance or survival, and ignore causal mechanisms that link landscape processes and land uses to habitat change or habitat change to fish population response. These ESU-wide assessments can indicate general patterns of population declines resulting from different land uses, but the data are generally too coarse to allow detailed analyses of habitat change and its effects on fish populations (e.g., Lunetta et al. 1997, Pess et al. 1999a). More detailed assessments for watersheds and individual populations involve collection of information on historical and current habitat abundance and quality, and therefore take more time to complete. These assessments can provide more detail on the types of habitat losses that have occurred, and which life stages might be most impacted for individual species or populations.

Phase II assessments identify where ecosystem processes or functions have been impaired, and therefore where ecosystem rehabilitation or restoration actions are needed. They do not focus on individual listed species, but focus on ecosystem attributes and processes that support multiple salmon species. ESU-wide correlation analyses of impaired processes and salmon populations are typically part of the Phase I coarse-resolution assessment described

above, and may require little additional work beyond reinterpretation of outputs from the Phase I analysis. However, some additional analysis of landscape or land use variables may be required to provide results that more specifically address changes in ecosystem processes. Watershed-level assessments in Phase II are more detailed and time-consuming, and managers should expect such that such inventories and assessments will take several years to complete. These watershed-level assessments have two essential components: 1) a screening component to identify where in the watershed each ecosystem process or function is most impaired, and 2) a field inventory component to identify specific actions that are needed to restore those processes. The screening assessments are primarily intended to help focus field inventories where they are most needed, but can also provide a general sense of where different types of recovery actions are likely to be focused and how much those actions might cost.

In recovery domains where a single species is listed, the restoration actions identified in Phase II assessments can later be prioritized using the life-cycle information from Phase I. In such cases, life cycle models indicate which habitats are likely most limiting, and therefore where and what types of restoration actions are likely to improve population performance. This approach alters the sequence of ecosystem recovery actions, but all actions remain focused on restoring landscape processes and functions that sustain salmon habitats over the long term (e.g., Beechie and Bolton 1999). The main risk inherent in this approach is that life cycle models provide only a “best guess” about which actions will most improve a population, and errors in the model may lead managers to focus too heavily on restoration actions that are not in fact limiting recovery of a population (see the “Managing Uncertainty in Habitat Recovery Planning” section, page XXX). Therefore, it is important to emphasize a number of ecosystem recovery actions simultaneously even where a single species is the focus a recovery strategy. Where it is

not appropriate to focus on any single species for recovery planning, other schemes for prioritizing actions should be employed (see the “Prioritizing Potential Restoration Actions Within Watersheds” section, page XXX).

Implementing Habitat Actions and Updating the Recovery Plan

The assessments described in this document are part of a larger sequence of steps needed to enact long-term strategies for salmon recovery. For most groups it will be many years before all Phase II inventories are completed, and it is not currently possible to reliably predict most population responses to habitat actions. Therefore, it is important to have a strategy for implementing interim actions, learning how populations respond to those actions, and updating the recovery plan as new information comes in. Recognizing the long-term nature of inventories, restoration experiments, and salmon recovery allows the tasks to be sequenced logically for implementation. In essence, the steps are:

1. Identify the restoration goals and objectives.
2. List assessments needed to identify appropriate habitat restoration actions (i.e., the assessments needed to complete Phase I and Phase II planning).
3. Identify and compile existing assessments to identify initial restoration actions.
4. Implement preliminary restoration actions as experiments, conduct remaining assessments and inventories to fill in the data gaps, and revise the plan and actions as new information comes in.

Step 1 describes the overall restoration strategy and types of information required to implement it. To some extent, the goals of restoration strategies for salmon recovery are constrained by the purposes of the ESA (i.e., conserve the ecosystems upon which listed depend) and the CWA (i.e., protect the biological integrity of aquatic systems). Consistency with the

purposes of these two acts will simplify the assessments needed to identify necessary habitat protection and restoration actions, and help avoid conflicts that arise from managing for the specific (and often conflicting) habitat requirements of multiple species. Examples of restoration goals are to “protect and restore the processes that form and sustain habitats to which salmonid stocks are adapted” (SWC 1998), “restoration and protection of habitat conditions and processes upon which the fish depend” (LCFRB 2001), or “to have a diversity of habitats and natural processes necessary to sustain healthy populations of native species” (Willamette Restoration Strategy 2001). Along with these habitat restoration goals, strategies should also have clearly stated near-term and long-term objectives. Near-term objectives should incorporate those actions that we already know are consistent with conservation of ecosystems that support salmon (e.g., removal of migration blockages, habitat protection through easements or purchase). Longer term objectives should include management experiments to clarify which actions are most beneficial to aquatic ecosystems and listed species, as well as implementing larger restoration projects that require changes in infrastructure or land uses (e.g., modifying levee systems to reopen access to estuary habitats). In short, strategies should describe how different actions will be identified and prioritized, how long-term inventories can be incorporated into the recovery plan, and how monitoring information will feed back into updates of the recovery plan.

Step 2 should explicitly list the types of assessments required to implement a recovery strategy. The complete recovery plan will include assessments for all of the Hs (habitat, hydropower, hatcheries, and harvest). This document provides guidance on assessments that will be useful in Phase I and Phase II planning for habitat actions (only one of the Hs).

Step 3 examines the list of needed assessments to identify information that already exists as well as those assessments that remain to be conducted. At this stage information can be

compiled to identify interim habitat recovery actions. Some examples of this stage of assessment can be found in the examples of OWEB Watershed Assessments or Washington Conservation Commission Limiting Factors Analyses (<http://www.oweb.state.or.us/publications/index.shtml>, <http://salmon.scc.wa.gov/reports/index.html>).

Step 4 includes two main components: 1) completion of inventories that identify specific habitat recovery actions (e.g., barrier inventories or riparian condition inventories), and 2) conducting restoration experiments to improve our understanding of which types of actions will most benefit salmon in each recovery domain or ESU. Inventory data can be directly incorporated into a recovery plan to expand the list of actions necessary to restore ecosystem processes and functions that support salmon recovery. Monitoring of management experiments improves our ability to predict the outcome of restoration actions, and can be used to adjust priorities in the recovery plan.

Table 1. Primary questions to answer in developing habitat recovery plans.

Question	Analysis area	Data types
<i>Phase I questions: Assessing changes in habitat availability and potential impacts on population characteristics</i>		
How might habitat changes have altered the abundance of individual populations?	ESU or Watershed	ESU: mainly remote sensing Watershed: mainly field
How might habitat changes have altered the population growth rate of individual populations?	ESU or Watershed	ESU: mainly remote sensing Watershed: mainly field
How might habitat changes have altered the diversity of life history patterns?	ESU or Watershed	ESU: mainly remote sensing Watershed: mainly field
How might habitat changes have altered the spatial structure of populations?	ESU or Watershed	ESU: mainly remote sensing Watershed: mainly field
What scenarios of habitat characteristics would support a viable ESU (viable meaning adequate levels of all 4 VSP parameters)?	ESU	Mainly remote sensing
<i>Phase II questions: Assessing disruptions to ecosystem functions and biological integrity</i>		
Where has biological integrity been degraded?	Watershed	Field
Where have watershed processes and ecosystem functions been impaired?	Watershed	Field/remote sensing

Table 2. Regional differences in dominant ecosystem processes or functions in the Pacific Northwest. This table is intended only to illustrate that different processes and assessments should be emphasized in different ecoregions. Important ecosystem processes vary within ecoregions, and watershed-level assessments should target those processes that are locally important within each watershed. (Note that the Columbia River estuary is in the coastal forests ecoregion, but also affects Columbia River stocks in the Western deserts and Western forested mountains.)

Watershed process or function	Level II ecoregion		
	Western deserts	Western forested mountains	Coastal forests
<i>Sediment</i>	Gullying and surface erosion (especially in agricultural areas)	Mass wasting and gullying	Mass wasting (surface erosion in agricultural lowlands)
<i>Flood hydrology</i>	Snowmelt dominated flood regime	Snowmelt dominated flood regime	Rain and rain-on-snow flood regime
<i>Low flow hydrology</i>	Diversions and dams common	Diversions common and dams	Diversions and dams less common
<i>Riparian functions</i>	Grasses and shrubs, some forest in floodplains	Sparse forests, shade a dominant function	Dense forests, wood recruitment a dominant function
<i>Habitat connectivity</i>	Culverts, dams, and dikes common; incision and floodplain abandonment common	Culverts, dams, and dikes common	Culverts, dams, and dikes common
<i>Estuary function</i>	NA (Columbia estuary should be assessed in relation to freshwater habitats)	NA (Columbia estuary should be assessed in relation to freshwater habitats)	Severe impacts in agricultural and urban areas
<i>Biological integrity</i>	Especially important in urban and agricultural areas	Especially important in urban and agricultural areas	Especially important in urban and agricultural areas

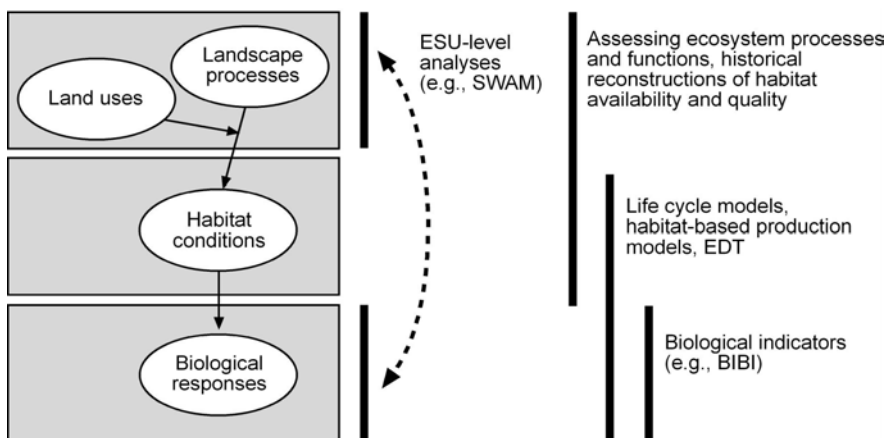


Figure 1. Schematic diagram of linkages among landscape processes, land use, habitat change, and biological responses (adapted from Beechie et al. in press). Assessing the biological response directly (e.g., using a biological indicator) identifies where ecosystem functions have been impaired, and may suggest causes of impairment. Assessments of habitat loss and resultant salmon population declines can be conducted by relating historical and current habitat abundance and condition to salmon utilization and survival. Assessing disrupted ecosystem functions and processes identifies causes of habitat change that result in diminished biological integrity and declines in salmon populations. For ESU-wide analyses of land use effects on salmon populations, landscape and land use factors can be correlated with indicators of population performance (e.g., correlation analyses).

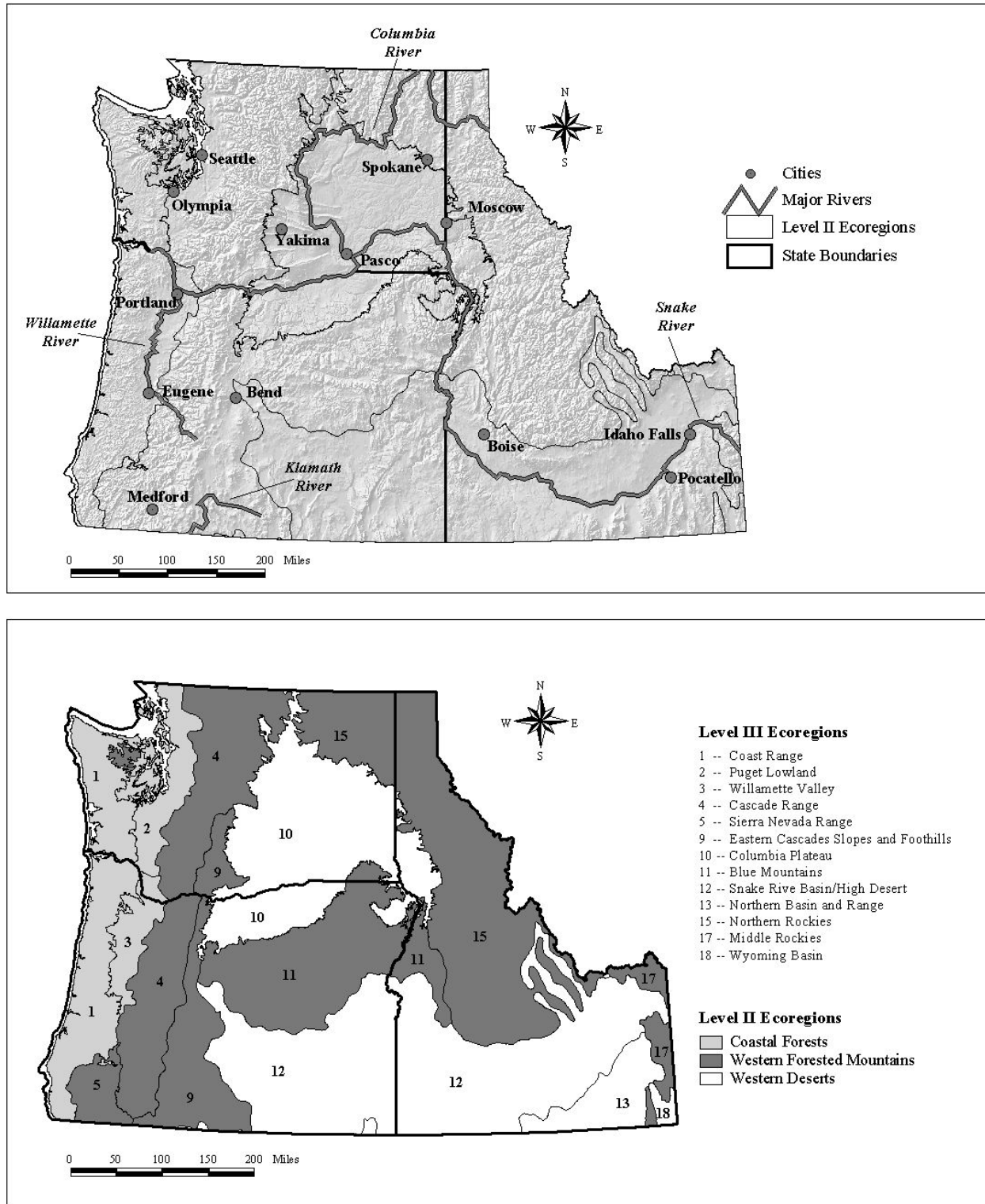


Figure 2. Level II and Level III ecoregions of the Pacific Northwest. (Note: ecoregion boundaries extend into northern California and other bordering states and provinces.)

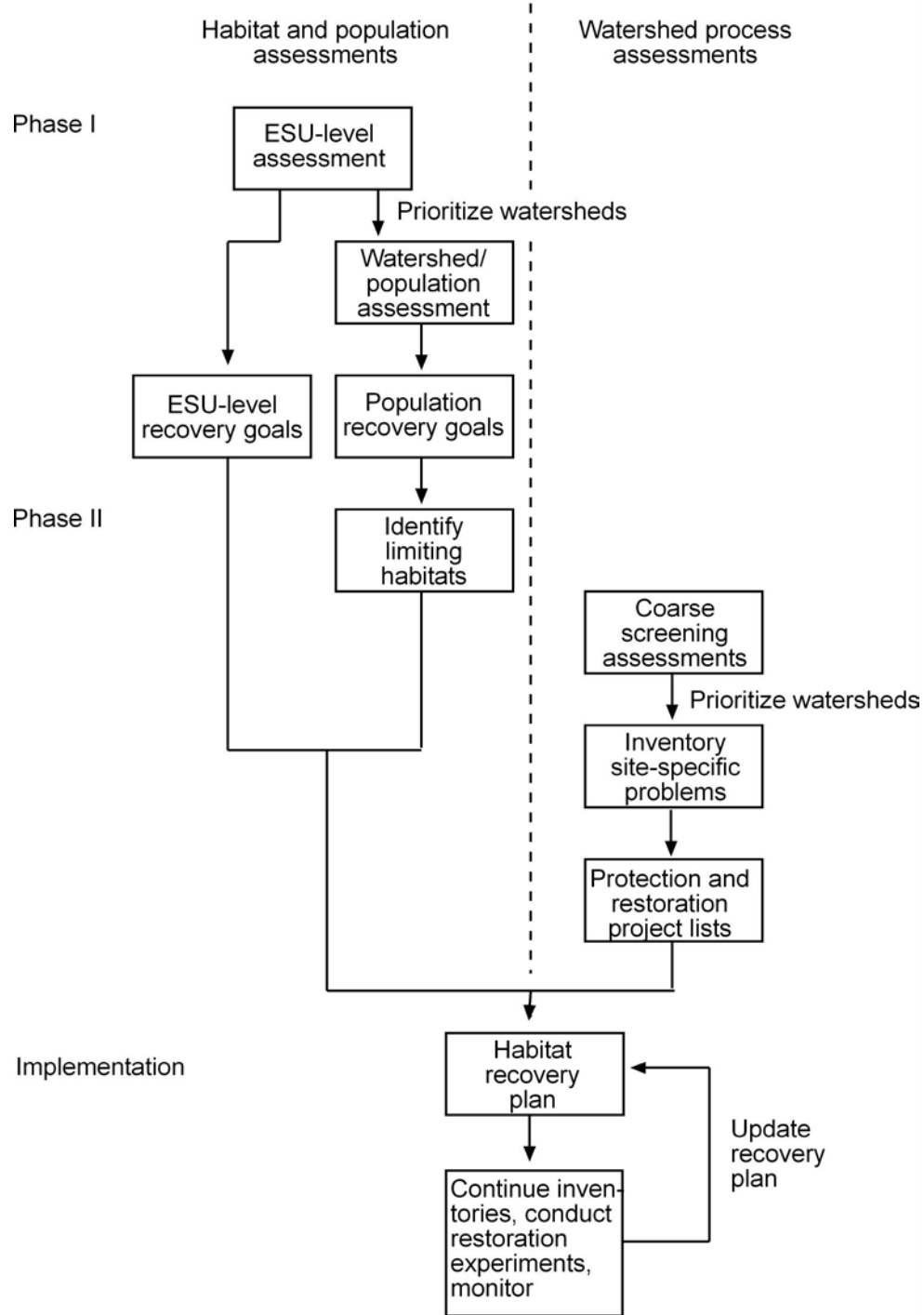


Figure 3. Generalized sequence for Phase I and Phase II assessments, integration of Phase I and Phase II information into the recovery plan, and plan updates based on continued inventories and monitoring.

ANALYSES FOR PHASE I RECOVERY PLANNING: SETTING RECOVERY GOALS

Recovery goals consist of a combination of quantitative and qualitative targets for fish populations, their habitats, and management outcomes (McElhany et al. 2000). Goals might include numerical fish population targets, numerical population trend targets, qualitative or quantitative targets for spatial distribution of populations and population diversity, habitat quality standards, or management outcome targets (McElhany et al. 2000). In this section, we consider how habitat analyses at the ESU and watershed scales can inform the development of viability goals for salmon populations and ESUs. These analyses also inform Phase II recovery planning by identifying factors causing decline and fish-habitat relationships. ESU-level analyses are used to examine the quantity and quality of habitat across numerous populations within ESUs, whereas watershed-level analyses focus on questions specific to individual populations.

As ESU and population viability goals are set, we must simultaneously evaluate whether current, historic, or “restored” habitat might be sufficient to support populations of the desired size, as well as whether the type and distribution of available habitats can support the desired diversity and spatial structure of salmon populations. Where current habitats cannot support the desired populations, we must identify and prioritize ecosystem restoration actions that will eventually help achieve the recovery goals (see the “Analyses for Phase II Recovery Planning” section, page XXX, and the “Prioritizing Potential Restoration Actions Within Watersheds” section, page XXX). For any of these analyses it is critical to select habitat measures that can be linked to population performance and are sensitive to land use changes or restoration actions. Habitat measures (physical, chemical, or biological) that meet these criteria facilitate an understanding of how land uses or restoration actions change habitats and how those habitat

changes in turn create population responses in salmon (see also Figure 1). This is true whether the analysis is conducted with coarse- or fine-resolution data available over entire ESUs, or only in certain watersheds.

Because habitats and biota are hierarchically structured, it is also important to view habitat data in the context of a hierarchical classification system such as illustrated in Table 3 (See also Appendix A, page XXX, for further discussion of scale issues in Phase I recovery planning). With such a classification of habitats, results of analyses across entire ESUs can be linked to fine-resolution analyses within individual watersheds. Moreover, this hierarchical structure allows one to construct simple predictive models for estimating abundance and distribution of fine-resolution habitats based on available coarse-resolution data (e.g., Lunetta et al. 1997).

ESU-level Analyses for Abundance and Survival Goals

ESU-level analyses differ from watershed-level analyses in that they ask questions and analyze data that span the area of entire ESUs. The area of existing ESUs ranges from 4,500-24,000 square miles. The number of independent salmon populations within each ESU varies from as few as one to more than 30 (see <http://www.nwfsc.noaa.gov/cbd/trt/>). Because these analyses encompass large geographic areas and often more than one salmon population, the goals and data needs are inherently different from efforts directed at watershed, stream, or reach scales.

The goal of an ESU-level analysis is to answer two of the recovery planning questions identified in Table 1:

1. How might habitat changes have altered the abundance, population growth rate, spatial structure and diversity of individual populations and ESUs?
2. What scenarios of habitat characteristics would support a viable ESU?

The nature of these questions, combined with the geographic scale of analyses, means that ESU analyses will typically correlate landscape characteristics (e.g., land use and land form) to characteristics of habitat and fish (Figure 1) in order to describe large-scale patterns of habitat change (Table 4). In general, fine-resolution data (e.g., habitat typing, barrier inventories, etc.) used in watershed-level assessments are not comprehensively available across the large geographic areas these analyses must cover. Therefore, ESU-scale analyses use currently available data (usually at coarse-resolution) to rapidly address questions that span large geographic areas (see Tables 4 and 5).

In this section we describe two approaches to assessing how land uses might be related to salmon abundance (or survival rates). The first uses simple correlations among landscape/land use variables and salmon abundance to evaluate the relative quality of different stream reaches within a large study area (e.g., Pess et al. 2002, Feist et al. in prep.). The second uses coarse resolution data to estimate historical and current habitat abundance within an ESU. Analyses for addressing changes in density independent survival rates (either for single life stages or for returning adults per spawner) will be similar to those for abundance except that survival measures will be used as the response variable rather than abundance.

Correlation Analyses

Correlation studies that link patterns of land cover and land use to measures of fish abundance, fish survival, or instream habitat quality are a first step toward understanding the ESU-level relationships between salmonid populations and the physical, chemical, and biological components of their habitat. Correlation analyses can utilize a broad range of metrics for population performance (e.g., genetic diversity, juvenile abundance, adult abundance, population growth rate, life-stage specific survivals), landscape characteristics (e.g., road density, geology,

land use), or habitat attributes (e.g., percent pools, water temperature, number or concentration of contaminants). These studies can be used to make predictions about where habitat conditions might limit or enhance salmon populations, to generate initial prioritizations of habitat action types and locations, to generate hypotheses for further testing, and to suggest important factors to control when setting up small-scale experiments, monitoring projects, or large management experiments (Figure 4). However, they cannot identify cause and effect relationships because of correlations among habitat descriptors, correlations among landforms and land uses, and the potential for unmeasured variables to explain existing patterns.

One example of this type of analysis is SWAM, a series of spatial and statistical analyses that relate salmonid population performance metrics in a particular basin to landscape and land use characteristics derived from existing geospatial data layers. This analysis identifies descriptors of landform and land use that are correlated with fish population performance in a given watershed. SWAM has been used in the Salmon River basin in Idaho (Feist et al. in prep.), the Snohomish River basin in Washington (Pess et al. 2002), and the Willamette River basin in Oregon (Steel et al. in prep.). In these three basins, SWAM linked indices of adult fish abundance (redd counts or adult fish counts at index sites) to multiple descriptors of landscape conditions across the entire watershed draining to the index reach. An alternate metric, conditions in the riparian area directly associated with the index reach, was also tested in the Salmon and Snohomish basins. Alternate population metrics such as juvenile abundances or life-stage specific survivals are possible wherever adequate data exists.

The spatial and statistical analyses involved in the SWAM approach are comprised of five steps. First, conceptual mechanistic relationships between landscape features and population abundance during all freshwater life history stages are identified from the literature

and from local habitat biologists. These conceptual relationships define the habitat data layers to be used as potential predictor variables. Second, spatial heterogeneity in the salmonid population data is examined to determine if certain areas in the basin consistently exhibit better population performance than other areas. Third, landscape data layers are overlaid with the geo-referenced fish abundance (e.g., redd counts) data. Fourth, statistical model is developed to describe annually consistent relationships between landscape characteristics and fish abundance. And finally, these relationships are rolled into a predictive model and applied to the entire basin of interest or to specific locations within the basin of interest (Figure 4). Many variations on the population and landscape metrics used in this type of analysis are possible. The best choice for a particular basin will be driven by available data.

SWAM or SWAM-like analyses provide a broad-brush estimation of current or potential fish occupancy within a basin and a first-cut estimate of the coarse-scale factors affecting abundance. Because SWAM is based on empirical data and a limited number of modeling steps estimates of prediction uncertainty is possible. In the Willamette basin, SWAM is being used to estimate potential fish occupancy behind barriers and then to prioritize barrier removal projects. Correlative models relating fish population performance to habitat conditions may also help identify the best remaining reaches or subwatersheds in a particular basin. If clear relationships between fish populations and habitat conditions exist, these analyses may suggest indicator habitat features and may identify areas that were historically productive. Ecological insights developed from these analyses may suggest likely habitat factors limiting population performance in a particular basin. Experience from these studies can be used to identify habitat characteristics to control when setting up experiments and monitoring an evaluation programs. Predictions of areas likely to support strong populations can suggest areas where detailed

watershed assessments and habitat inventories should be conducted (see the “Habitat Analyses for Phase II Recovery Planning” section, page XXX).

Current and Historical Potential Habitat

A second approach for evaluating the ability of multiple watersheds to support viable salmon populations is to examine the distribution and quantity of current and historic habitat across large geographic areas. This approach evaluates associations between fish and specific types of habitat for multiple watersheds and populations (e.g., comparing 20 demographic populations). These analyses use limited habitat survey or historical distribution information; ancillary topographic and hydrologic spatial data are used to supplement empirical reach-level data, and ultimately provide a tool for estimating habitat-based fish densities for multiple watersheds. In addition, these analyses may rely primarily on geospatial data sets such as digital terrain models, Landsat imagery, and regional hydrography to predict physical stream features (e.g., Lunetta et al. 1997). The widespread availability of corresponding spatial datasets (remotely sensed imagery, Geographical Information System or GIS data, and spatially-explicit modeling methods) permits the derivation of representative reach-level geomorphic and hydrologic information for multiple watersheds (Table 5). This information can be used in conjunction with field-based mapping data, such as anthropogenic modifications (e.g., Bortleson 1980), to refine estimates of currently and historically available habitat for different species. Subsequently, methods to refine historic habitat estimates can be used to estimate possible levels of historical fish densities across the watershed.

A specific example of this type of assessment is an inventory of currently and historically available stream reaches, and the amount of habitat blocked by migration barriers such as dams, diversions, and culverts (e.g., Steel and Sheer in prep.). Where inventories of migration barriers

are available they can be used to address several questions for ESUs and their constituent populations.

1. How many miles of stream habitat are unavailable or unreachable to fish due to the construction of anthropogenic barriers?
2. What proportion of historic habitats are currently useable for juvenile and adult salmonids?
3. Is current habitat sufficient to support viable salmonid populations?

In Phase II recovery planning for identifying and prioritizing actions, these same analyses can provide additional information toward developing a list of actions (e.g., which barrier removals might provide the greatest habitat and population benefits).

The first step of the analysis includes stream network acquisition or generation and barrier identification. Modeled stream networks may be used to achieve the appropriate analysis scale and multiple barrier data layers will likely need to be examined for accuracy and combined. By comparing historically available habitats to those currently available below migration barriers, these analyses describe the degree and type of habitat loss for each population. For example, mapping of 2,600 barriers in the Willamette/Lower Columbia (WLC) ESU and the amount of stream blocked by each shows that certain demographic populations have been more strongly influenced by man-made barriers than other populations (Figure 5).

Assessments of whether accessible or inaccessible stream habitats are or might be useful to fish requires a classification of more detailed information about reach-level stream characteristics (e.g., stream gradient, drainage area, or channel width). The classification scheme can be based on existing models, published literature and/or interviews with local biologists, and reach attributes may be available from field data, estimated from aerial photos or modeled. In

the WLC, stream reaches in specific gradient ranges were identified as possible or prime spawning and rearing habitat for winter steelhead trout (*Oncorhynchus mykiss*), summer steelhead, chinook salmon (*O. tshawytscha*), and chum salmon (*O. keta*) based on a survey of local biologists, and these gradients were modeled for the WLC domain using a procedure developed by Miller (2003) that employs a digital elevation model to generate stream reach information (e.g., Figure 6).

The remaining question is whether the amount of useable habitat is sufficient to support VSPs across the ESU. In the Puget Sound and WLC ESUs, demographic models were used to estimate draft goals for individual populations (see <http://www.nwfsc.noaa.gov/cbd/trt/>). The amount of useable habitat calculated as described above was used to verify whether target numbers are realistic given the actual proportion of stream miles likely to be used by salmon, either currently or historically. Fish densities implied by a range of population abundance goals (goals divided by total number of prime or possible stream kilometers) can be estimated and evaluated (see Steel and Sheer in prep. for additional details). Because the inventory approach is based on multiple GIS datalayers, each with variable precision and accuracy, as well as an estimated classification system, the precision and accuracy of the resulting estimates should be assessed (e.g., by comparing predictions to field measures as in Lunetta et al. 1997). Continued work on this topic will make the inventory approach more useful for decision-making.

Watershed-level Analyses for Abundance, Survival, and Population Growth Rate Goals

For Phase I recovery planning, watershed-level analyses are designed primarily to assist in setting abundance goals for recovery, although they can also shed light on goals for density independent survival rates, population growth rates, life history diversity, and spatial structure.

These watershed-scale analyses provide greater detail on how habitat changes might have altered salmon populations than do ESU-level analyses. This additional detail is important in salmon recovery planning for several reasons. First, landscape, land use, and habitat differences among watersheds prevent generalizations about limiting factors, potential abundance, and habitat capacity. Second populations are locally adapted to watershed-level conditions, so habitat needs of salmon populations may vary among watersheds. Third, the types of habitat changes that cause changes in salmon fitness and survival may differ among watersheds. Finally, watershed-level analyses answer questions that require fine-scale field data that is only available for particular watersheds.

In order to conduct these analyses it is important to consider how different analysis approaches represent the salmon life cycle. In a generalized salmon life cycle model (Figure 7), the number of surviving fish at the beginning of each life stage is a function of capacity and density independent survival in the previous stage. Both capacity and density independent survival are affected by habitat quantity and quality, and the number of smolts per spawner represents the combined life stage transitions from spawning to salt water entrance. Habitat analyses at the watershed-level are conducted using one of three general approaches: 1) simplified limiting factor models, 2) complex expert system models such as EDT for evaluating limiting habitat conditions, and 3) dynamic life cycle models to estimate population responses over time. All approaches are based on the salmon life cycle, and all assess historic and current habitat conditions in a watershed to estimate how habitat changes may have altered salmon abundance or survival at different life stages. However, the approaches differ in two main respects.

First, each approach emphasizes different aspects of the life cycle and different parameters driving stage-to-stage survivorship. Limiting factors models such as the coho salmon (*O. kisutch*) limiting factors model (Reeves et al. 1989) focus on changes in capacity at each freshwater life stage and treat density independent stage-to-stage survivals as constants. (Density independent survival here refers to survival at low population size, Figure 7). EDT allows both capacity and density independent survival to change at each life stage, but does not estimate population response over time. Therefore, both of these approaches can help with setting abundance goals, but EDT can also help assess goals for life stage survivals or returning adults per spawner at low population sizes. The dynamic life cycle model allows both capacity and survival to vary at each life stage, and also explicitly evaluates population growth or decline over time. Therefore, it can be used to help evaluate population growth rate goals in addition to abundance and survival goals.

The second main difference among the approaches is in data requirements. EDT is a relatively rapid assessment technique that relies on expert judgments of historic and current habitat conditions for more than 40 individual habitat variables requiring inputs for each reach and each month of the year, as well as estimated relationships between habitat and fish survival. The empirical limiting factors and life cycle approaches focus on fewer parameters that can be measured or estimated from empirical relationships, and require longer implementation time due to the time required to gather habitat data.

In this section, we review the three approaches to Phase I analyses at the watershed-level in the context of setting abundance goals for recovery. For this discussion we group the limiting factors and dynamic life cycle models together because they both rely on similar reconstructions

of historical habitat availability. EDT is considered separately because of its reliance on expert judgment for most of its habitat input data.

Ecosystem Diagnosis and Treatment

EDT is a complex salmon production model that has been used throughout the Pacific Northwest for salmon recovery planning. The conceptual basis for EDT is described in Lichatowich et al. (1995) and Mobrand et al. (1997), and the details of the model structure and its use are found in Lestelle et al. (1996). EDT is a habitat-based model that relies primarily on expert opinion (best professional judgment) as environmental input data. It organizes environmental inputs, estimates habitat condition, and predicts fish population performance based on estimated life-stage specific stock-recruit functions. Fish population performance is characterized using a “survival landscape,” which is created by multiplying the life-stage functions together. The survival landscape is typically created for environmental states corresponding to two points in time, current and historic, but may also be created for other states. The spatial unit of analysis is a reach (e.g., one river mile to several river miles).

There are three levels of information in an EDT analysis. Level 1, the input data, includes 45 environmental variables (or some subset thereof), including empirical data (e.g., data measured in the field), derived data (e.g., estimated from coarse-scale data), and anecdotal information (e.g., best professional judgment). Level 2 is the set of biological rules that drive the model. Biological rules are working hypotheses about how a salmonid species at a given life stage will respond to a specific environmental variable. The shape of each biological rule is estimated from published literature and expert judgment. The level of proof behind each rule is ranked based on the information underlying it. The sum of the functions results in a combined estimate of the survival of fish throughout their life cycle. Level 3 is the translation of the rules

into 17 biological performance attributes (variables) that give relative survival or performance by life stage. Each attribute is used to modify an idealized or optimum condition that is the “historic template.” Survival and performance by life stage are the sum of the 17 biological attributes, and are always a fraction that ranges between 0 and 1.

EDT has been used to estimate habitat capacity under current, historic, and restored conditions. These have been proposed as the basis for population viability abundance goals. EDT can help organize existing expert opinion and empirical data to create hypotheses about linkages among certain habitat conditions and fish abundance. However, because the underlying data and functional relationships are largely untested, the accuracy of any EDT outcome is unknown. As noted in Mobrand et al. (1997), this performance measure is an indicator of how favorable the environment is (or might become) for salmon to persist and abound, not a predictor of how many will return and when. Currently, EDT does not provide estimates of the uncertainty (or precision) of the output estimates. Output from EDT will be more useful for decision-making if estimates of both accuracy and precision can be generated.

Estimating Current and Historical Potential Fish Production

The simplest approach to estimating how habitat changes have altered fish abundance at the watershed level is to use a simple limiting factors models to assess current and historical habitat availability and production potential (Reeves et al. 1989, Beechie et al. 1994). Many studies have quantified juvenile and adult fish use of particular habitat types (e.g., Bisson et al. 1988, Groot and Margolis 1991, Lichatowich 1999, Montgomery et al. 1999), and general patterns of habitat use by different species are largely consistent among studies (Figure 8, Pess et al. in press). Therefore, it should not be necessary to develop new fish-use relationships for each

watershed and limited field data should be sufficient to confirm that existing data from nearby streams can be used.

Unlike associations among fish densities and habitat types, habitat data are not transferable across watersheds. The natural potential of stream networks and the effects of land use on habitat condition vary by watershed (Frissell et al. 1986, Lunetta et al. 1997, Beechie et al. 2001, Collins and Montgomery 2001). Therefore, habitat inventories must be conducted separately in each river basin or planning area. Assessments for estimating historical and current habitat in a particular watershed are conducted in two steps: 1) estimate historical and current abundance of habitat types, and 2) estimate population responses (simplified limiting factors model or dynamic life cycle model).

Assessing changes in habitat availability

A habitat classification system suitable for estimating historical and current habitat and potential fish production must have two main attributes. First, analysts must be able to associate fish abundance and survival with each habitat type. Second, to estimate changes in potential production over time, it must be possible to quantify both historical and current areas of each habitat type. We recommend a suite of habitat types at two hierarchical scales, similar to that shown in Table 3. The coarser resolution of habitat types can be mapped from remotely sensed data at the reach scale (e.g., topographic maps, aerial photography, or satellite information), whereas the finer resolution of habitat types must be identified in the field at the habitat-unit scale (sometimes with the aid of aerial photography). Because these typing systems are nested, all reaches within a watershed can be stratified by landscape and land use factors using the remotely sensed coarse-resolution data, and reaches within each strata can be subsampled to develop distributions of habitat types within each reach type. This hierarchical system enables extrapolation of habitat conditions for unsampled reaches within the watershed. Stratification of

reach types may include several different landscape and land use factors, although a relatively small number of strata are desirable to reduce the complexity and number of assumptions and calculations. For example, tributary reaches may be stratified simply by slope and land use in order to identify changes in pool area as a result of land uses (Beechie et al. 2001). However, the same slope classes are not particularly relevant for large rivers, where some combination of slope and discharge may be more useful in predicting natural channel patterns (e.g., Leopold et al. 1964).

Methods for estimating current and historical habitat abundance differ among habitat types. Therefore, it is not possible to describe a single methodology for assessing changes. Instead, we provide an overview of different approaches that one might use for assessing habitat present and historical conditions (Table 6) and references that provide greater detail on specific methods.

Assessing population responses

Once habitat changes have been quantified, changes in potential population sizes can be estimated for specific life stages using two basic approaches: a simplified limiting factors model (e.g., Reeves et al. 1989) or a dynamic life-cycle model (e.g., Emlen 1995, Botsford and Brittnacher 1997, Ratner et al. 1997). As described earlier, the limiting factors approach focuses only on juvenile life stages, and allows habitat capacity to vary while treating survival parameters as constants. By contrast, the dynamic life cycle model can incorporate changes in capacity or survival, and evaluates population responses over long time frames.

Simplified limiting factors analysis—As described in Reeves et al. (1989), smolt production potential from a given habitat type or area is calculated as:

$$\text{habitat area} \times \text{average fish density} \times \text{survival to smolt.} \quad (1)$$

A comparison of the impact of habitat alterations on smolt production potential requires separate estimates for each habitat type. Thus, the production potential of a habitat for each life stage (e.g., spawning, egg to fry, summer rearing, winter rearing, smolt migration) can be expressed mathematically as:

$$N = \left(\sum_{i=1}^k \left(\left(\sum_{j=1}^n A_{ij} \right) \times d_i \right) \right) \quad (2)$$

where $\sum A_{ij}$ is the sum of areas of all habitat units ($j=1$ through n) of type i , and d_i is the density of fish in habitat type i (Beechie et al. in press). To compare capacities among life stages and identify which habitats may be limiting smolt production, the population estimate (N) for each life stage in a given habitat is multiplied by density independent survival to smolt stage so the capacities can be compared in terms of number of smolts ultimately produced (Reeves et al. 1989). Equation 2 can also be used to estimate historical spawner capacity (or potential population size at other life stages) based on estimates of historical habitat availability (Appendix B, page XXX). Both spawning and rearing capacities can then be used to develop or assess population viability abundance goals and can be incorporated into assessments of factors that limit population size.

Dynamic life cycle modeling—This is a third tool for evaluating how habitat changes affect salmon abundance and population trends (growth or decline), using the same historical and current habitat data as above. A typical form of habitat-based life cycle model is the Leslie matrix (e.g., Emlen 1995, Botsford and Brittnacher 1997, Ratner et al. 1997), which can consider how fish move from one habitat to another and can employ density dependent or density independent relationships to describe transitions between stages (Greene and Beechie, in prep.).

Thus, each life stage transition within this model can be governed by capacity, survival, or movement parameters.

As with the preceding approaches, there are few estimates of stage-to-stage survivals for use in such models, and there are few data to describe the form of relationships among habitat and life stages (e.g., density independent mortality, density dependent mortality, or density dependent movements to less favorable habitats). However, this approach forces analysts to specify how mortality and movements are governed between life stages, and to address uncertainties stemming from lack of data. Therefore, one important use of such models is to begin evaluating which assumptions and data most strongly affect model outputs so that recovery goals can be set with appropriate caution.

In one example of this approach, Greene and Beechie (in prep.) found that predicted improvements in Puget Sound chinook salmon population growth rate or escapement strongly depend upon habitat-specific survival and residency estimates, as well as knowledge of density-dependent mechanisms. Both population growth rate and escapement showed the greatest sensitivity to nearshore and ocean survival regardless of the existence and mechanism of density dependence (Figure 9). However, delta conditions had the next greatest effect on salmon abundance (after nearshore and ocean survival) under density dependent survival, whereas stream conditions had the next greatest effect under “hockey stick” and density-dependent movement scenarios. In addition, simply altering the density dependence assumptions while holding all habitat variables the same altered predicted abundance of chinook salmon by more than a factor of four (Figure 10).

Such findings have direct implications for setting recovery goals, as well as for identifying habitats that may limit recovery of populations. First, these models point out that model

assumptions and parameters can dramatically alter our predictions of population responses to habitat changes, and therefore that abundance goals should be set with these uncertainties in mind. Second, such models can identify parameter and model uncertainties (see the “Managing Uncertainty in Habitat Recovery Planning” section, page XXX) that substantially alter our conclusions about which habitats limit recovery, and therefore shift the focus of restoration efforts. Finally, such a model can indicate that certain life stage survivals (e.g., nearshore for ocean-type chinook salmon populations) may be most sensitive to habitat change regardless of the assumptions about density dependence. However, a sensitivity analysis simply begs the question whether significant habitat changes have occurred and whether restoration is feasible. Therefore, such analyses also motivate efforts to characterize poorly understood habitats, their effects on salmon abundance and survival, and the extent to which they have been modified.

Analyses for Spatial Structure and Diversity Goals

Both coarse and fine-scale (see Table 3) comparisons of current and historic habitat within a watershed can indicate whether the diversity of habitat types available to salmonids today are vastly different from the past, whether connectivity between habitat types has been altered, and subsequently whether life-history diversity or spatial structure have been altered. While changes in distribution and diversity of habitats can be identified through these analyses, more detailed work is needed to address patch size and natural dynamics within individual systems. In this section, we describe existing approaches to relating habitat conditions to population structure and diversity at multiple scales and suggest how this type of information might be used to inform the development of recovery goals.

ESU-level Analyses

Few studies have tried to determine how changes in habitat may have altered the diversity and spatial structure of salmon populations and ESUs (Table 4). However, several studies have examined genotypic and phenotypic diversity at a variety of scales, and from these studies Healey and Prince (1995) concluded that the majority of genotypic diversity is contained within stocks while most phenotypic diversity is greater across populations and landscapes. This suggests that ESU-level analyses may find stronger relationships among habitat variables and diversity than will watershed-level analyses.

Several large-scale analyses for salmon have been completed, and they indicate that certain habitat changes have radically altered the spatial structure and population diversity of several ESUs. For example, the Interior Columbia Basin Ecosystem Management Project (Quigley and Arbelbide 1997) characterized the distribution of individual fish species, patterns in fish species diversity and status, and their relationships to landscape characteristics. Their analyses indicate major declines in the number and distribution of salmonid species located in Columbia Basin subwatersheds. Anadromous fish species have been largely extirpated from large portions of their range, and the consequences of this for ESU viability need to be addressed.

In another example of large scale analysis, Waples et al. (2001) characterized patterns of intraspecific diversity for Pacific Northwest salmon species along three major axes: ecology, life history, and biochemical genetics. The ecology axis included characteristics of freshwater habitats that are of known importance to salmon (i.e., hydrography, temperature, vegetation, geology, etc.). The results indicated that the amount of diversity within a species was significantly related to the extent of ecological diversity experienced by that species. For

example, diversity measures were lower for pink salmon (*O. gorbuscha*) and chum salmon, which are limited in distribution to areas directly affected by Pleistocene glaciation. In contrast, chinook salmon and steelhead trout were broadly distributed and exhibited the greatest degree of diversity. Results such as these indicate that reductions in species ranges will likely alter spatial structure and diversity within ESUs, and may ultimately constrain recovery of some ESUs under certain management scenarios.

Frissell (1993) also mapped region-wide patterns in fish diversity using data compiled for inland fishes (Williams et al. 1989) and anadromous stocks (Nehlsen et al. 1991), focusing on the Pacific Northwest and California, and on species and stocks at risk. Mapping units were based on a drainage basin size of 50-2,000 km² and the basins categorized according to the number of species (0-1, 2-3, 4-5, or 6-8) classified as extinct, endangered, or threatened. Isopleths were then fitted between the categories. Results indicated a general increase in endangerment from north to south and identified basins in Idaho, Puget Sounds, and Northern California in which population declines were mainly due to large-scale dams and irrigation projects. Dams, logging roads, and floods have damaged habitats and severely threatened faunal diversity in other basins in Southern Oregon and Northern California.

Watershed-level Analyses

Smaller scale analyses are rare and their paucity suggests that we do not yet understand the mechanisms by which local habitat conditions affect the diversity and spatial structure of fish populations. One analysis that attempts to bridge this gap is an analysis of bull trout (*Salvelinus confluentus*) by Dunham and Rieman (1999) within the Boise River. Their results indicate that the large-scale geometry of catchments can influence the distribution of aquatic species. Bull trout populations in larger, less isolated, less disturbed patches are more likely to persist. They

conclude that disturbance within these habitats should be minimized and they speculate that conservation and restoration opportunities might best be centered within those patches of intermediate size or isolation.

Quantifying relationships between habitat and population diversity and identifying the mechanisms by which species are distributed across the landscape and utilize habitat patches are necessary steps in the recovery planning process. We need a better understanding of how these processes affect population persistence and extinction risk. Information generated from analyses at a wide range of scales must be integrated into ESU-level goals for population diversity and spatial structure. Population or watershed specific targets can then be developed.

Conclusions

Habitat analyses at multiple scales are necessary for setting goals related to abundance, population growth rate, spatial structure and diversity. These large and small-scale analyses compare current and historic habitats to evaluate whether current habitat is sufficient to support viable salmon populations and ESUs. ESU scale analyses apply consistent techniques to examine questions over large spatial scales, often relying on remotely sensed information and coarser resolution habitat data. Watershed scale analyses focus on questions relevant to individual populations.

The analyses described and discussed in this section are designed to answer questions necessary to set biological recovery goals (Phase I). As these recovery goals are set, the next phase of recovery planning must address how to reach those goals. The data and information collected for the analyses can also contribute to identification of factors causing decline, quantification of fish-habitat relationships, and efforts to develop prioritized lists of actions for

ESUs and watersheds. The following sections will examine these questions pertinent to Phase II recovery planning in more detail.

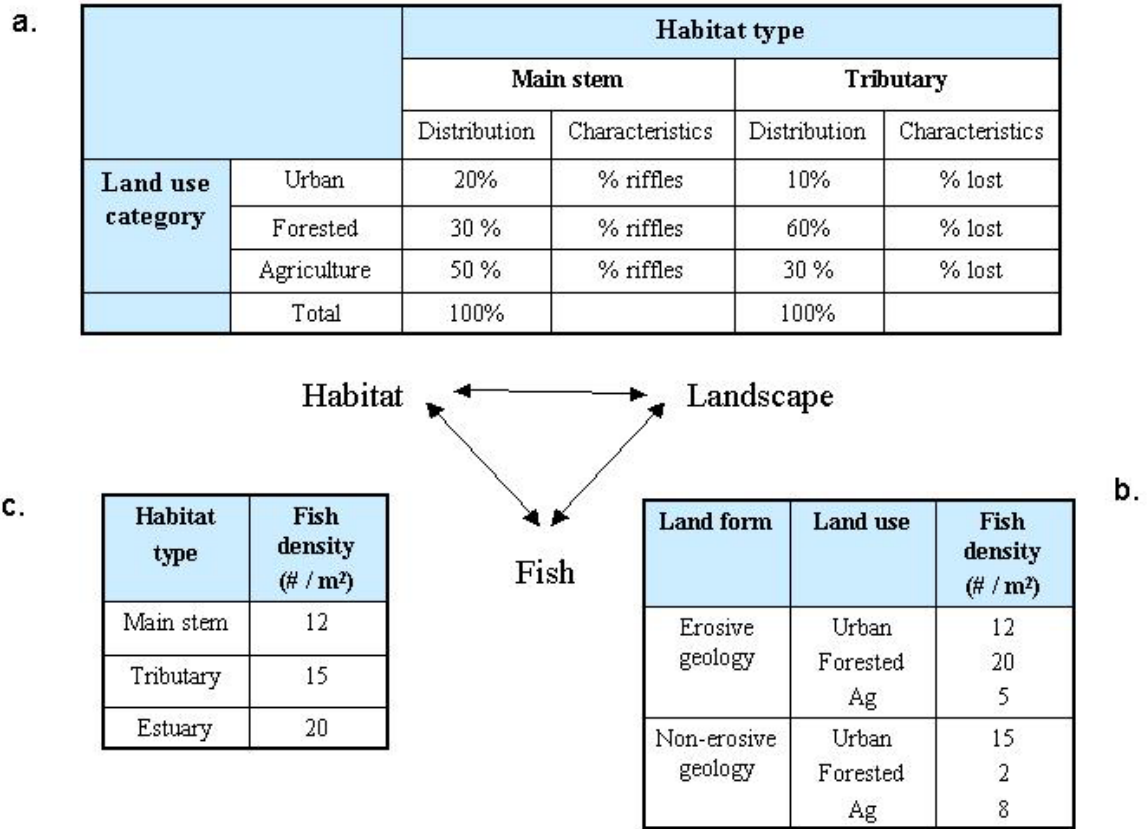


Figure 4. Illustration of the kinds of relationships between habitats, landscape, and fish that broad-scale analyses can examine: Table A illustrates how habitat quantity and characteristics are distributed across different land uses, Table B illustrates how fish density in three land use categories differ in erosive vs. non-erosive geologic settings, and Table C describes how fish density differs across habitat types. The numbers in the tables are completely fictitious and for illustrative purposes only.

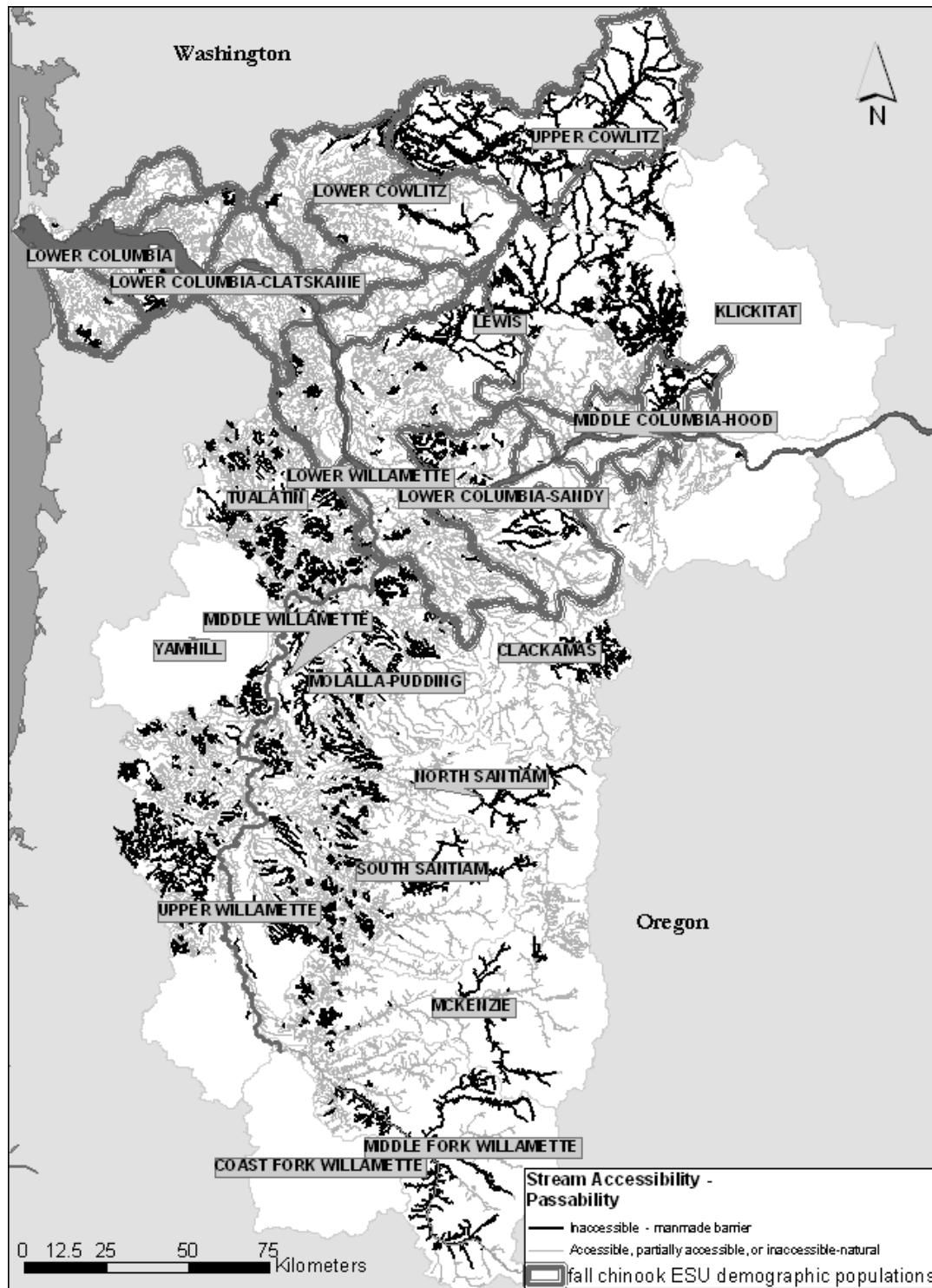


Figure 5. Stream accessibility and passability for all streams considered in the WLC analysis. Legend describes categories of accessibility. Stream km that are inaccessible because of man-made barriers are indicated in black. Labels indicate the fourth field hydrologic basin, and thick gray outlines indicate the boundaries fall chinook demographic populations for the Lower Columbia fall chinook ESU.

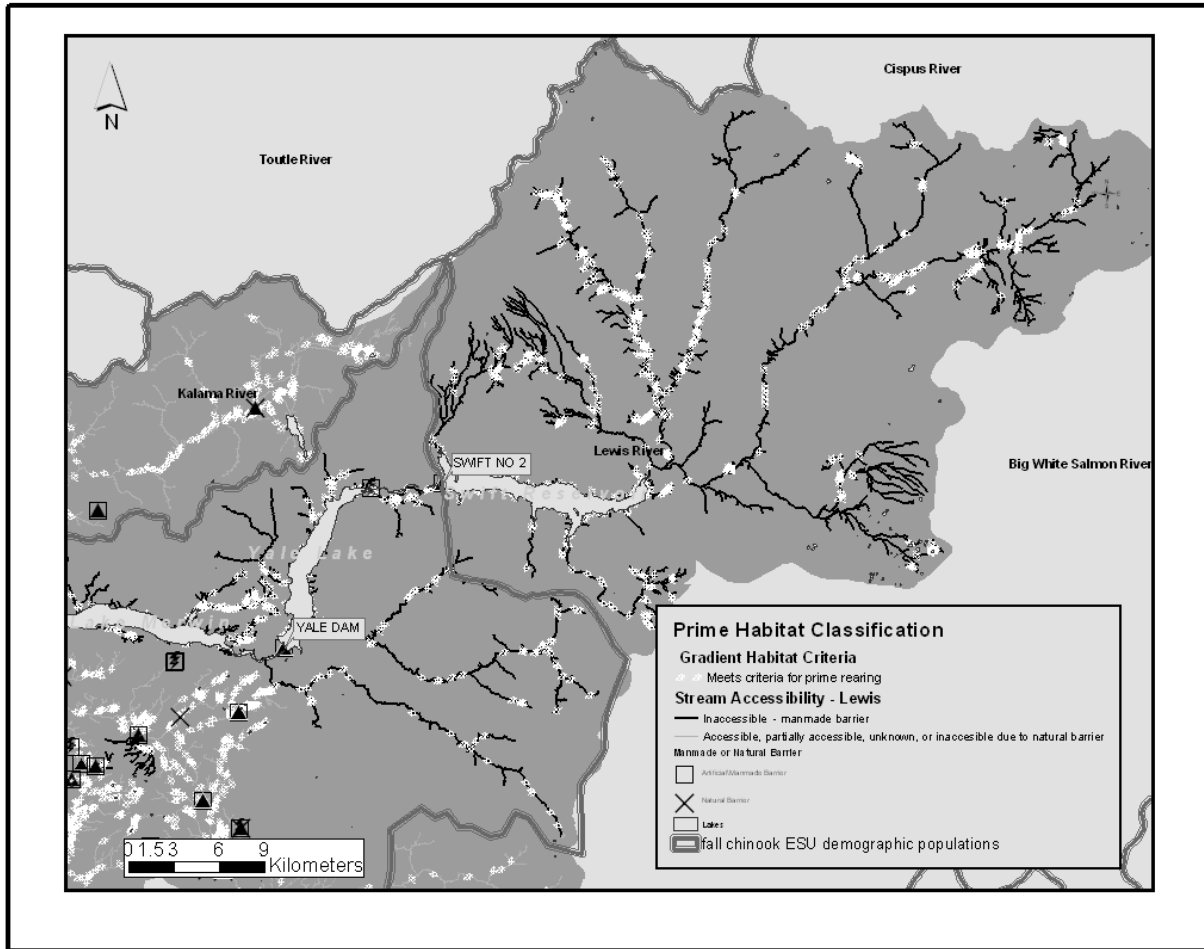


Figure 6. Example of the identification of prime and possible habitat attributes. Map indicates stream reaches classified as “prime habitat” for chinook rearing or spawning in the Lewis River, based only on defined gradient thresholds. The white symbols indicate patches of streams (reaches) that meet the thresholds. Black lines represent streams inaccessible to fish due to man-made barriers. Thick gray lines represent the boundaries of fall chinook demographic populations included in the Lower Columbia fall chinook ESU.

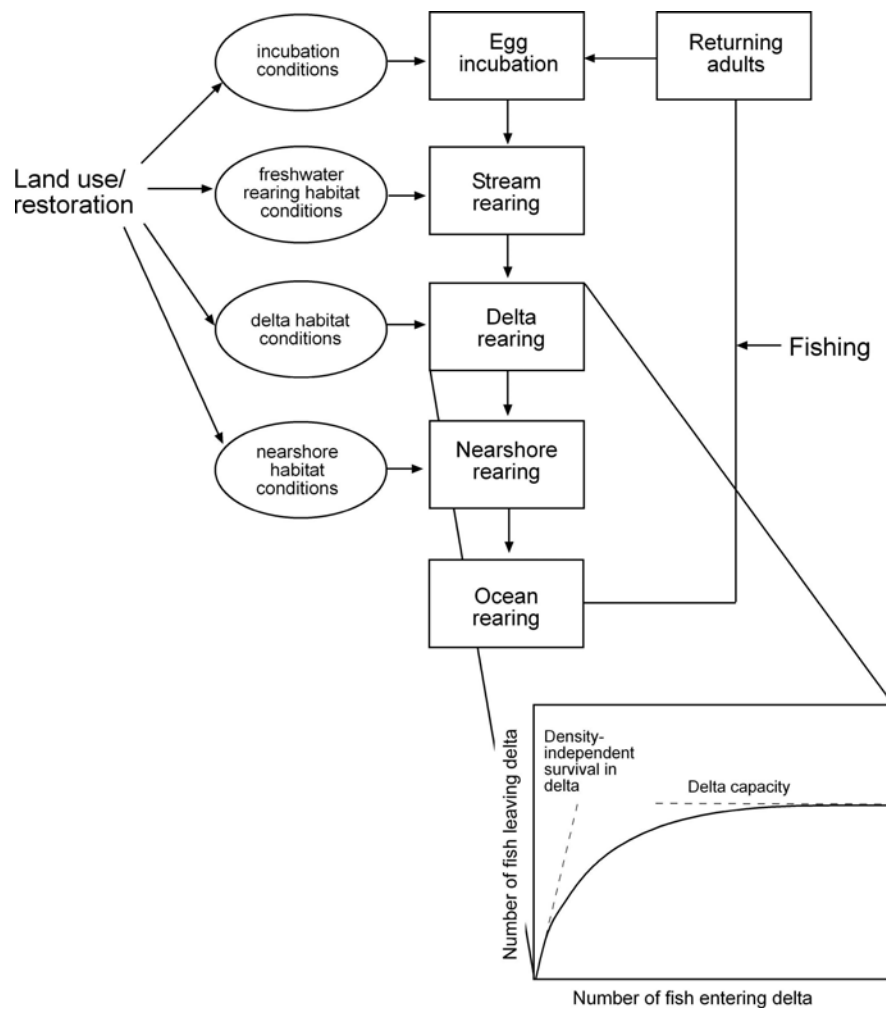


Figure 7. Generalized life cycle model illustrating linkages among life stage transitions (boxes), habitat conditions affecting those transitions (ovals), and human actions altering habitat conditions or survival. Expanded panel illustrates a typical density dependent relationship for one life stage, with density independent mortality at low population size and increasing density dependence as population size approaches habitat capacity. (Note, however, that all life stages may not be regulated by density dependant mechanisms.)

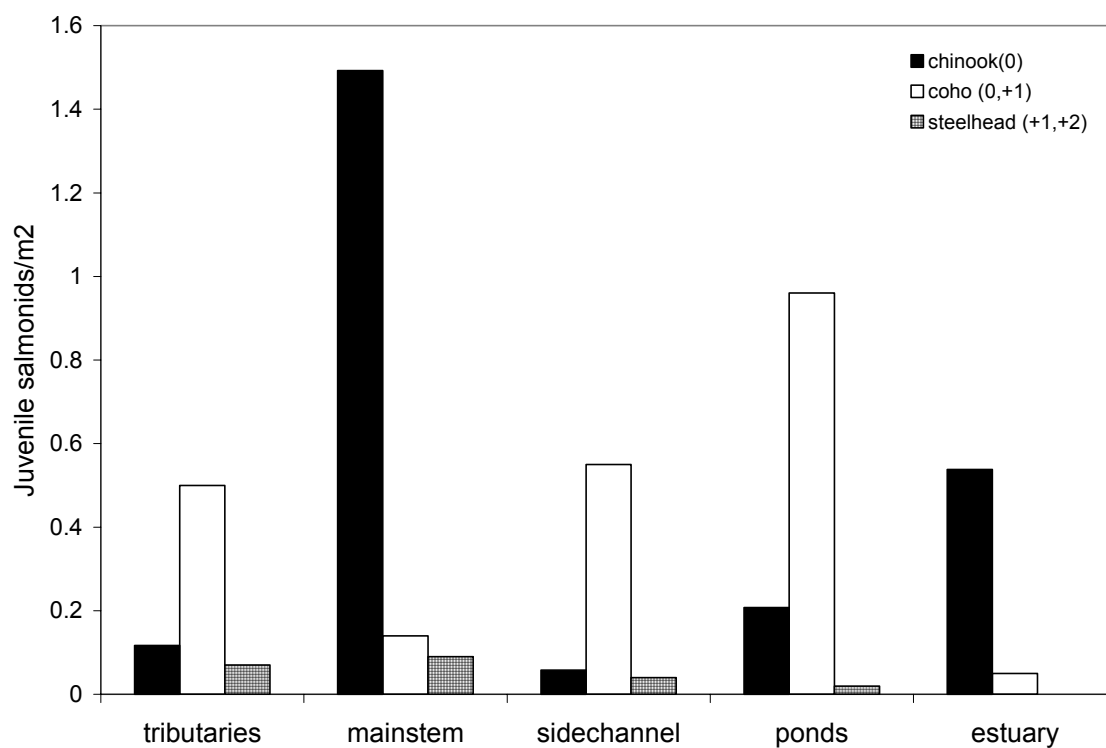


Figure 8. General juvenile salmonid use at the habitat scale. Compilation of over 60 references.

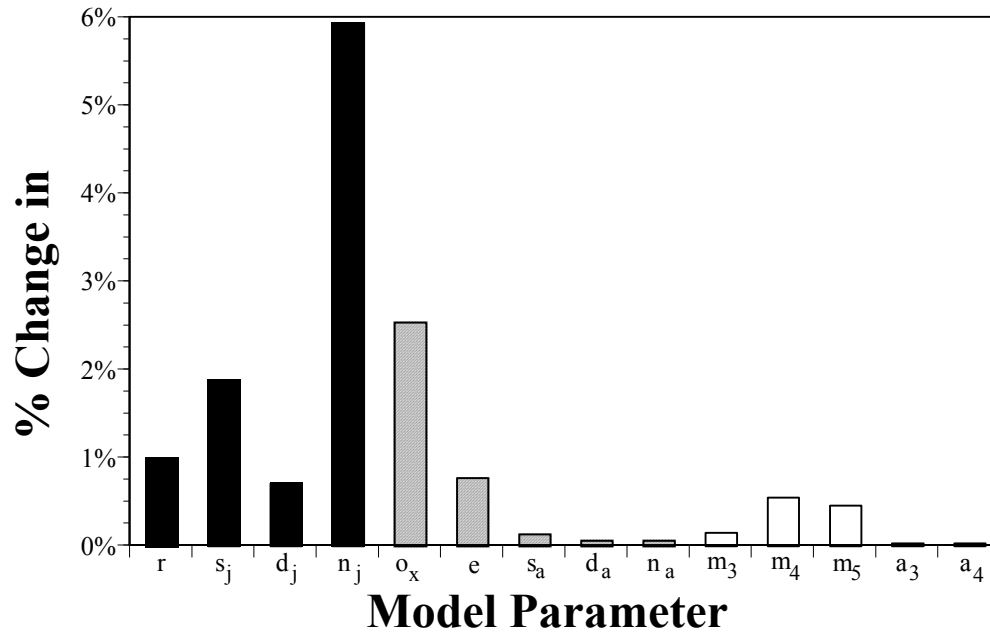


Figure 9. Sensitivity analysis of lambda (λ) to changes in model parameters when population dynamics of Puget Sound ocean-type chinook salmon are density-independent. Each bar represents a percentage change in λ resulting from a 5% change in each particular model parameter. Black bars are juvenile habitat parameters, hatched bars are subadult and adult habitat parameters, and open bars are reproductive parameters. Parameters are: r = redd survival, s_j = juvenile stream survival, d_j = tidal delta survival, n_j = nearshore survival, o_x = annual ocean survival, e = escapement, s_a = adult survival back through the stream, d_a = adult survival back through the tidal delta, n_a = adult survival back through the nearshore, m_x = age-specific fecundity, and a_x = age-specific breeding propensity. (From Greene and Beechie in prep.)

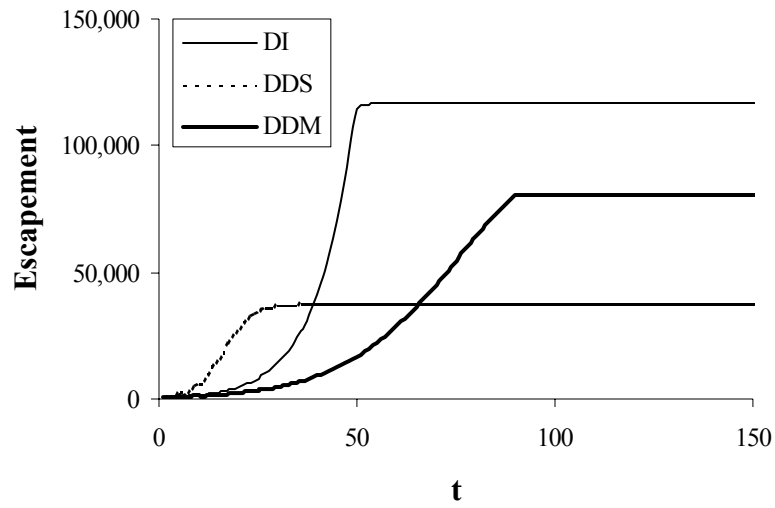


Figure 10. Predicted Puget Sound ocean-type chinook salmon escapement as a function of time in years (t) for density-independent (DI), density-dependent survival (DDS), and density-dependent movement (DDM) scenarios with a spawning capacity of 60,000. (From Greene and Beechie in prep.)

Table 3. Habitat types used for the two types of watershed assessments described in this paper. Coarser-scale habitat types are mapped from topographic maps, aerial photography, and satellite imagery. Finer-scale habitat types are mapped using a combination of aerial photography (for larger units) and field measurements. Adapted from Beechie et al. (in press).

Habitat type (Coarser scale) ←		→	Habitat type (Finer scale)
Large main stems (>50m bfw) by channel type based on gradient and confinement	<ul style="list-style-type: none"> • Mid-channel • Edge 		<ul style="list-style-type: none"> • Mid-channel pool • Mid-channel glide • Mid-channel riffle • Boulder/cobble • Cobble/gravel • Bar edge • Bank edge • Natural • Hardened • Backwater (alcove)
Small main stems (10-50m bfw) and tributaries (<10m bfw) by channel type based on gradient and confinement	<ul style="list-style-type: none"> • Pools • Riffles 		<ul style="list-style-type: none"> • Pool • Scour • Plunge • Trench • Backwater • Glide • Run • Rapid • Riffle
Off-channel habitat within large main channel floodplains	<ul style="list-style-type: none"> • Channel-like • Pond-like 		
Impoundments			<ul style="list-style-type: none"> • Ponds < 500 m² • Ponds > 500 m² and < 5ha • Lakes > 5 ha
Palustrine wetland	<ul style="list-style-type: none"> • Forested • Scrub/shrub 		Open water area by season
Riverine tidal wetland	<ul style="list-style-type: none"> • Forested • Scrub/shrub 		Open water area by season and tidal stage
Tidal-delta wetland	<ul style="list-style-type: none"> • Scrub/shrub • Emergent 		Open water area by season and tidal stage
Tidal-delta channel	<ul style="list-style-type: none"> • Mainstem • Blind • Distributary 		

Table 4. List of analyses that address questions pertaining to the four categories of recovery goals for salmon.

Recovery goal	Examples of analyses	References
Population abundance and growth rate	Quantification of current vs. historic habitats	McIntosh et al. 2000, Thompson and Lee 2000, Thurow et al. [year?]
	Effects of irrigation diversions in the Salmon River Basin	McClure et al. in press
	Land-use impacts on salmon	Bradford and Irvine 2000, Thompson and Lee 2000, Paulsen and Fisher 2001
Spatial structure and diversity	Quantification of current vs. historic habitats	Thurow et al. [year?]
	Regional patterns in fish diversity	Frissell et al. 1993b, Waples et al. 2001
	Associations between fish assemblages and habitat	Waite and Carpenter 2000
	Presence/absence mapping	Quigley and Arbelbide 1997

Table 5. General description of potential analytical approaches for estimating historical and current habitat abundance at the ESU level. Note that methods are not readily available for estimating historical conditions for most habitat types. Thus, ESU-scale analyses most commonly focus on direct loss of important habitat types.

Type of habitat change	Possible analysis methods
Tributary and mainstem blockages	Utilize digital information on known barriers and historical ranges (e.g., Quigley and Arbelbide 1997). May also model historical ranges based on channel slopes and stream size [reference?].
Channel type (e.g., based on Montgomery and Buffington 1997)	Estimated from vegetation information, hydrography and digital elevation models (Lunetta et al. 1997).
Off-channel, wetland or beaver pond areas	No remote sensing methods readily available.
Lakes	No remote sensing method available.
Beaver ponds	No remote sensing method available.
Estuary	No remote sensing method available. Use existing information where possible (e.g., Bortelson et al. 1980).

Table 6. General description of analysis approaches for estimating historical and current habitat abundance at the watershed-level. References provide more detail on specifics of methods.

Habitat type	Analysis methods	References
<i>Reduced off-channel or wetland areas</i>	Historical habitat areas estimated from historical maps, notes, and photos, and often field verified by residual evidence of their prior locations. Present-day areas can be measured from aerial photographs and in the field.	Beechie et al. 1994, Collins and Montgomery 2001.
<i>Lakes</i>	Changes to lake areas are measured directly from historical and current maps and typically indicate where rivers have been dammed for hydropower or water supplies.	Beechie et al. 1994
<i>Beaver ponds</i>	Pre-settlement beaver pond areas estimated based on frequencies of beaver ponds in relatively pristine areas, or predictive methods using stream and valley characteristics. Present-day pond areas within the study area measured using field surveys and aerial photography.	Naiman et al. 1988, Pollock and Pess 1998
<i>Tributary and mainstem blockages</i>	Portions of tributaries that are no longer accessible to salmon can be mapped using inventories of habitat upstream of migration barriers. Natural barriers to salmon migration must first be identified to delineate the assessment area. Habitat areas upstream of each man-made barrier must be surveyed to determine how much habitat is inaccessible.	Beechie et al. 1994, WDFW 1998, OWEB 1999a, Pess et al. 1999b
<i>Altered pool abundance</i>	Based primarily on data from reference sites within the study area, but may also use historical information where available.	Beechie et al. 1994, WDFW 1998, OWEB 1999a, Nickelson and Lawson 1998, Collins and Montgomery 2001

ANALYSES FOR PHASE II RECOVERY PLANNING: IDENTIFYING ECOSYSTEM RESTORATION ACTIONS

In this section we describe assessments and inventories to identify ecosystem protection or restoration actions within individual watersheds during Phase II of recovery planning. These assessments focus on how land uses have disrupted ecosystem processes and functions that support salmon, and identify specific actions needed to correct those problems. At the ESU-level, analyses described under Phase I identify broad habitat types that are most impaired within each watershed (e.g., estuary, mainstem, tributary), as well as broad categories of habitat recovery actions (e.g., barrier removals). We do not repeat discussion of these assessments here, but instead focus on assessments and inventories that develop site-specific lists of ecosystem protection and restoration actions for individual watersheds.

Within individual watersheds, there are two types of assessments that can be used during Phase II recovery planning. The first type includes screening assessments that are used in watershed-level analyses to focus inventories on areas where they are most needed. The second type includes field inventories that identify specific locations where habitat protection or restoration actions are needed (see also Figure 3). Screening assessments can rely largely on existing data (e.g., geospatial coverages, hydrologic data, existing inventories) to map impaired processes, but may also include field assessments of biological integrity to help identify potential ecosystem disturbances where causes of habitat degradation are more complex (e.g., urban areas). By contrast, specific projects (such as a culvert barrier to fix or a riparian area to be fenced) can only be identified with more specific field data. Both types of assessments are relevant in all ecoregions, although specifics of the assessments and inventories may vary depending on which ecosystem processes are most important in different ecoregions.

Earlier we presented a conceptual diagram illustrating how watershed controls and natural landscape processes combine to form habitat conditions (Figure 1). In this section we expand the conceptual diagram to explain linkages among ultimate controls, proximate controls, habitat-forming processes, habitat conditions, and biological responses (Figure 11). Ultimate controls are independent of land management over the long term (centuries to millennia), act over large areas ($>1 \text{ km}^2$), and shape the range of possible processes and habitat conditions in a watershed (Naiman et al. 1992, Beechie and Bolton 1999). Proximate controls are affected by land management over the short term (i.e., years to decades), and act over smaller areas (Naiman et al. 1992).

For organizational purposes, we group ecosystem processes and functions into three categories: 1) distributed watershed processes (similar to non-point sources, such as supplies of sediment and water), 2) reach-level processes that primarily affect the adjacent reach (e.g., riparian processes and functions), and 3) other ecosystem functions (e.g., habitat connectivity). Ecosystem processes are typically measured as rates and characterize what ecosystems or components of ecosystems do (SWC 1998). For example, sediment or hydrologic processes in a watershed may be characterized by the rates (volume/area/time) at which sediment or water is supplied to and transported through specific locations of a watershed. Certain riparian functions such as wood recruitment to streams can be viewed similarly. By contrast, other ecosystem functions or attributes such as habitat connectivity are not well described as processes. Rather, migration barriers reduce the habitat capacity of a system, and flow diversions may divert or pump fish into irrigation systems. Neither fits neatly into the categories of watershed- or reach-level processes, and therefore are considered separately.

Watershed-level Analyses

The purpose of the watershed-level assessments is to identify: 1) the natural landscape processes and functions in a watershed, 2) the effects of land use on natural processes, and 3) the causal relationships between land use and changes to habitat conditions. Ecosystem restoration and protection actions resulting from these assessments are directed at protecting and restoring beneficial habitat-forming processes instead of attempting to build specific habitat conditions (FEMAT 1993, Spence et al. 1996, Moore 1997, Beechie and Bolton 1999). As described earlier, the watershed-level assessments systematically identify land use disruptions to habitat-forming processes at two levels of resolution. First, screening assessments locate disturbed habitat-forming processes using existing GIS data and limited field measurements for ground-truthing (e.g., Lunetta et al. 1997, Quigley and Arbelbide 1997, our Appendix C). These assessments identify where processes are most disrupted and focus field inventories on areas that are most in need of restoration. Second, field-based inventories identify specific alterations to flow regimes, sediment supply, riparian conditions, habitat connectivity, water quality, and channel and floodplain interactions (e.g., OWEB 1999a, our Appendix C). This more detailed assessment relies solely on field-based inventories and identifies specific restoration or protection actions that are required for recovery.

Assessing Degradation of Ecosystem Processes and Functions

Ecosystem processes and functions to inventory should include (at a minimum) hydrology, sediment supply, riparian functions, channel-floodplain interactions, habitat isolated from salmon access, and water quality (Table 7). This suite of inventories is based on current scientific knowledge of their effects on salmonid habitat and survival of salmon in freshwater, as well as knowledge of how various land use practices affect the processes. The list may not

include all impacts to salmon in a watershed, but it includes those that are clearly supported by scientific literature (e.g., Meehan 1991, WDNR 1995, OWEB 1999a) and that are responsible for a significant proportion of the total loss in salmon production from Pacific Northwest river basins. As described earlier in this report, different watershed controls and natural landscape processes will dominate in different ecoregions, leading to differences in the assessments to be conducted for identifying ecosystem restoration actions (Table 2).

Watershed-level processes

Watershed level processes are those that have multiple, widely distributed sources, including sediment supply, hydrology, and inputs of nutrients or pesticides (Table 7). Describing how these processes have been disrupted and what restoration actions are required for their recovery requires two different kinds of assessments. First, process assessments identify the degree to which process has been altered by land use, and where in each watershed these changes have occurred. Second, inventories are required to identify where specific restoration actions must be taken in order for recovery to occur.

Identifying altered sediment supply—Sediment supply to streams is altered by many processes including changes in mass wasting due to logging and road building (e.g., Sidle et al. 1985), increased surface erosion after prescribed burns (e.g., Megahan et al. 1995), increased surface erosion from unpaved road surfaces (e.g., Bilby et al. 1989), and surface erosion and gullying after grazing (Platts 1991, Elmore 1992, Johnson 1992, Trimble and Mendel 1995). We present two main approaches to understanding disruptions to these sediment supply processes: 1) budgeting and 2) landscape indicators based on known relationships of certain water and land uses to the parameter in question. The budgeting approach is often used for sediment supply, but can also be used for inputs of nutrients or pesticides to water bodies. The general budget can be stated in equation form:

$$\Delta S = I - O$$

Where, ΔS is change in storage, I is input, and O is output (e.g., Reid and Dunne 1995). In essence, S is the stream condition for any parameter (e.g., the amount of sediment or of a pesticide in the stream), and quantifying changes in inputs or outputs indicates how land uses have altered the stream ecosystem. In many cases, it may only be necessary to quantify how inputs have been altered by land uses, which is called a partial budget. That is, where changes to outputs are negligible, an increased input is approximately equal to the change in storage and to the altered stream condition. Therefore, it is not necessary to understand output processes in detail (e.g., sediment transport) in order calculate change in storage and to understand how the stream ecosystem has been altered.

We illustrate the partial budget approach by describing how one might approach two different altered sediment inputs to streams due to land uses: 1) extrapolation of limited empirical data for estimating increased fine sediment supply from tilled croplands in the Blue Mountain Level III Ecoregion, and 2) an empirical approach to estimating changes in landslide rates due to forestry activities in the North Cascades Level III Ecoregion. Both approaches focus on identifying where sediment supplies to streams have been significantly altered, and can help focus restoration efforts on areas that contribute large amounts of sediment. Note that these approaches do not identify the exact locations of necessary restoration actions, which may require specific inventories of croplands where eroded sediments deliver directly to streams or road segments with high risk of landsliding. For efficiency, these inventories can initially target those areas of high sediment supply identified by the partial sediment budgets.

The partial sediment budget for croplands makes use of measured erosion rates from soils with and without cover crop. Soils without cover crop erode at rates as much as 10 times higher

than that of soils with cover crop (see complete overview of processes and rates in Dunne and Leopold 1978), and varies with soil types, rainfall, slope, cover type, and other factors. Local erosion rates for different soils and cover crops may have been measured in many areas by researchers and land management agencies, and examples for different soils and cover crops can be found in Dunne and Leopold (1978). From known rates, a simple cumulative model for basins with can be expressed as:

$$I_c = (A_c \times E_c) + (A_{nc} \times E_{nc})$$

where I_c is the current total estimated sediment input to selected reach, A_c is the area of land with cover crop or other vegetation, E_c is the erosion rate per unit area with cover crop, A_{nc} is the area of land with no cover crop, and E_{nc} is the erosion rate per unit area without cover crop. A natural background rate of sediment supply from surface erosion can be estimated by applying the natural erosion rate to the entire basin:

$$I_n = (A_c + A_{nc}) \times E_{nc}$$

Calculations of I_c/I_n for various subwatersheds within a region can then be compared to identify those areas where tilling is likely to have significantly altered sediment supply to streams.

Where soil erosion rates have not been measured, it may be necessary to use predictive equations such as the Universal Soil Loss Equation (Wischmeier and Smith 1965, Wischmeier and Smith 1978) or Revised Universal Soil Loss Equation (Renard et al. 1991, Renard et al. 1997). Dunne and Leopold (1978) provides a good overview of the equation and its application, along with charts and tables for estimating certain parameters in the equation, and the original handbooks can be consulted for greater detail on the methods. Estimates of local soil parameters are generally available from local USDA Natural Resources Conservation Service offices.

In the second example, a partial budget for sediment supply in coastal forests is constructed by conducting landslide inventories from historical aerial photographs and estimating contributions of fine sediments from road surface erosion (e.g., Paulson 1997). The general approach is similar to that for the cropland erosion example, where the final product of the partial sediment budget is I_c/I_n and comparisons among basins indicate areas where sediment supplies have been altered significantly. In these inventories landslides are enumerated and measured on each aerial photograph, and volume of each landslide is calculated based on a relationship of photo-measured area to field-measured volume for a subset of the recent landslides. Land use association is also recorded for each landslide (e.g., clear-cut, road, or mature forest), allowing estimation of the aggregate impact of land use on the sediment input, as well as identification of the land uses most responsible for changes in sediment supply. Surface erosion estimates can be based on characteristics of road surfaces, cut and fill slopes, and precipitation (e.g., WDNR 1995). Calculation of I_n typically assumes that there is no surface erosion (overland flow and gullying are rare in the Coastal Forest ecoregion), and the landslide sediment production rate for mature forests can be applied to the entire basin. These sediment budgets can then be compared to identify areas where sediment supplies have been most altered and where modified timber harvest practices or road modifications can have the greatest impact on ecosystem recovery.

Landscape and land use indicators of altered sediment supplies can be developed from sediment budgets or field studies of erosion in order to more rapidly screen large areas for disrupted sediment supply. For example, GIS maps of geology, soils and hillslope angles can identify areas that are prone to landsliding (e.g., Montgomery and Dietrich 1994, our Appendix C), and overlays of land cover and roads can be used to identify where landsliding has likely

increased (Appendix C). Results of such screening analyses can then be used to identify areas where inventories of potential restoration actions should be focused.

In general, remote sensing methods and mapping of landscape indicators are used to identify areas for passive restoration, and field inventories are required to identify active restoration projects. For mass wasting dominated areas, mapping of landslide hazard areas is used to identify areas that are particularly prone to landsliding, and sensitive to land uses such as clear-cut logging or road building (Figure 12). Such maps identify passive restoration actions (e.g., areas to avoid for future logging), which allow recovery of sediment supply rates by preventing or modifying land uses within hazard areas. Second, inventory of road landslide hazards identifies specific areas for active restoration. Road inventories should identify segments of road that are at risk of failure (e.g., Renison 1998), as well as specific stream crossings, cross drains, or fills that are likely to fail. Each potential failure site can be itemized on project lists for restoration action. These restoration actions can then be prioritized based on protection of refugia, potential impact to stream habitat, smolt production, cost, and other factors.

Where surface erosion and gulying are dominant processes, terrain and soil conditions that lead to severe erosion can also identify areas where modified agricultural practices can reduce erosion and sediment supply to streams. Modeling of altered sediment yields by various conservation practices can help identify the types of agricultural practices that are greatest contributors to erosion (Williamson et al. 1998) as well as those restoration actions that are most likely to be successful (Ebbert and Roe 1998). These actions can also be itemized and prioritized based on a number of factors including the magnitude of reduced erosion, costs, and other factors.

Identifying altered hydrologic regime—Hydrologic processes can be altered by land uses in a variety of ways, including increased peak flows from impervious surfaces (e.g., Booth and Jackson 1997), livestock compaction (Trimble and Mendel 1995), increased peak flows from increased snow accumulation and melt (e.g., Zeimer 1981, Harr 1986, Beschta et al. 2000), and decreased peak flows or low flows from dams and withdrawals (e.g., Richter et al. 1996, Donato 1998, Spinazola 1998). Assessments of increased peak flows typically utilize landscape indicators of changes to watershed processes based on known functional relationships between land cover and peak flows (e.g., Booth and Jackson 1997, Beschta et al. 2000, our Appendix C), and may include detailed models that project changes in peak flow hydrographs as a result of land cover changes (e.g., Booth and Jackson 1997). Assessments of low flow changes typically include inventories of total withdrawals and calculation of the proportion of stream flow removed (e.g., Donato 1998, Spinazola 1998), as well as indirect estimates based on power consumption at pumping stations (e.g., Maupin 1999).

Known relationships among zoning, impervious surface area, and changes in hydrologic processes and biota have been used to indicate changes in hydrologic regime in urban areas (see Appendix C). Where impervious surface areas are less than 3% of the watershed area, hydrologic regime is not significantly different from one with no impervious surfaces (Booth and Jackson 1997). However, where impervious surfaces are more than 10% of the watershed area, hydrologic regime has likely been altered to the point where changes in biota are severe (Luchetti and Furstenburg 1992, Moscrip and Montgomery 1997, May et al. 1997). Similar coarse analyses can be developed for other watershed processes such as contaminant runoff from agricultural or urban areas, where high concentrations of compounds that are toxic to biota, such as salmon, alter their behavior in ways that could reduce survival (Scholz et al. 2000). More

general landscape indicators, such as percent of a watershed in urban land cover, may be a greater predictor of biological condition because it is a more inclusive measure of anthropogenic disturbance and not just a change in one specific watershed process such as hydrologic regime or water quality (Karr and Chu 2000, Morley and Karr 2002). Examples of use of land cover indices for peak flow changes from rain on snow are also included in Appendix C.

Indicators of hydrologic alteration (IHA) can be used to assess the degree of hydrologic alteration within a watershed (Ritcher et al. 1996). The method summarizes complex hydrologic variation using 32 stream flow parameters that have biologically relevant attributes (Ritcher et al. 1996). The hydrologic data is from known sources such as stream gages and wells. Each parameter is broken up into pre- and post-impact time frames, and the central tendency (defined as the mean and median) and dispersion (defined as the variance and coefficient of variation) are compared to assess degree of hydrologic perturbation (Ritcher et al. 1996). Perturbation can include activities such as dam operations, flow diversion, groundwater pumping, or intensive land use (Ritcher et al. 1997). The tool is to be used with other ecosystem metrics (e.g., biological integrity indices) and can help set stream flow based restoration targets, identify areas of hydrologic alteration, and measure progress towards quantified conservation goals (Ritcher et al. 1996, 1997, 1998). The tool is not meant to predict biological response to hydrologic alteration (Ritcher et al. 1996).

For alteration of low flows by water storage and diversions, data availability varies between large dams and small private irrigation or water supply diversions (Spence et al. 1996, Quigley and Arbelbide 1997). For large dams (greater than 2 m in height) inventory data are available for regional characterization of water withdrawals, and cumulative withdrawals from large dams can be calculated (e.g., Quigley and Arbelbide 1997). However, data for smaller

diversions and their effect on stream flows are less readily available (Spence et al. 1996), and inventories of low flow changes are needed to systematically identify stream reaches where low flows are at issue within individual watersheds.

Two types of assessments can be used to inventory low-flow impairments within watersheds and identify potential restoration actions. First, low-flow impairments can be identified through inventories of diversions and quantification of water withdrawals. The existing 303d database from the U.S. Environmental Protection Agency (EPA) identifies more than 13,000 km of flow-impaired stream reaches in the interior Columbia Basin (Quigley and Arbelbide 1997) as well as additional streams in western Washington and Oregon. This database can be used as a preliminary inventory of withdrawals and low-flow impairments. More detailed field inventories of withdrawals can be conducted within watersheds to list all small dams, diversions, and pump stations that withdraw water from streams and assess the degree to which stream flows are reduced. State water rights databases (e.g., Washington Department of Ecology, Oregon Water Resources Department) provide a starting point for such assessments, although field inventories of actual withdrawals are often needed to confirm which water rights are currently in use. In general, ranking of the proportion of stream flow withdrawn in various reaches indicates which reaches deviate most significantly from natural stream flows, and are most likely in need of increased stream flows. Methods available to identify specific locations where water withdrawals impact stream flows include source metering using stream flow gauges or other measurement devices (e.g., Donato 1998), direct assessment of stream flow change (e.g., Ritcher et al. 1996), and estimates of water withdrawal based on power consumption at pumping stations (e.g., Maupin 1999).

In a similar fashion, identifying recovery actions for alterations to peak flow hydrology include both protection and restoration actions. In areas where hydrologic regime approximates the natural regime, ecosystem management should focus on protecting current hydrologic processes. These actions might include avoiding additional hydrologic changes by preventing new impervious services and forestry impacts to peak flows. By contrast, where hydrologic regime deviates significantly from the natural regime natural, restoration actions should be identified. For alterations to peak flows, identification of restoration actions may include actions to alleviate impervious surfaces (Holz et al. 1998, Maryland DER 2000) and to alleviate impacts of clear-cuts and roads on peak flow responses. For low flow impairment, identification of restoration actions is perhaps more problematic as many withdrawals are unlikely to stop completely. Instream Flow Incremental Methodology (IFIM) methods or variations of it as well as flood recurrence information can be used to help identify how much stream flow is necessary to support aquatic ecosystems at both low-flow and high-flow periods (Jowett 1997).

Identifying altered water quality—Water quality parameters can also be used to indicate areas with a high likelihood of disruption, especially with regard to temperature and nutrient or pesticide inputs. Again, the EPA 303d list provides a useful starting point for identifying disruptions to water quality. Many streams throughout the west are listed as water quality impaired, which indicates that some type of restoration may be necessary. In general, further field inventories may be necessary to clarify the exact nature of the problem, and then to identify corrective actions.

Water quality assessments may include direct measures of water quality parameters, and relationships among the parameters and biotic assemblages may help identify where disruptions are most important (Waite and Carpenter 2000). For example, multivariate classification and

ordination were used to examine patterns in chemical and physical variables in association with relative fish relative abundance in the Willamette River Basin (Waite and Carpenter 2000).

Patterns of fish assemblages were primarily related to water temperature, dissolved oxygen, and stream channel gradient at the ecoregion scale, however, chemical concentrations of pesticides and total phosphorus were more important than physical habitat features in low gradient floodplain ecoregions such as the Willamette Valley (Waite and Carpenter 2000). Water quality restoration actions may thus need to be a priority in such areas. (Additional biological indicators are discussed later in this section.)

Reach-level processes

Reach-level processes are those processes that directly affect the adjacent reach (Table 7). These processes mainly include riparian functions and floodplain-channel interactions. An extensive body of literature describes linkages between riparian forest functions and stream habitat, which in turn affect the survival and abundance of salmonids. Riparian functions include supply of wood and leaf litter to streams (Naiman et al. 1992), shading (Beschta et al. 1987), root reinforcement of stream banks and floodplain soils (Platts 1991, Elmore 1992). Dominant functions vary by ecoregion, although many streams even in the driest ecoregions have or had a forested riparian corridor (Platts 1991). Channel and floodplain interactions form a wide array of habitats that salmonids historically occupied (Sedell and Luchessa 1982, Peterson and Reid 1984, Collins et al. 2002). Many of these habitats are now either destroyed or inaccessible to salmon due to the effects of levees, dams, channel incision, or other land uses (Sedell and Luchessa 1982, Beechie et al. 1994, Peacock 1994, Shafroth 1999, Pohl 1999, Beechie et al. 2001, Collins et al. 2002).

Identifying disrupted riparian processes—The level of wood input or other riparian functions increases with increasing width of forest buffer on streams (Figure 13), and the proportion of the

function occurring within a given distance of the channel edge varies by function (e.g., Sedell et al. 1997). These types of relationships can be used to evaluate the current status of functional interaction between a stream reach and riparian area, and indicate whether existing levels of riparian protection are sufficient to ensure continued function. For this example we focus on recruitment of wood to streams for forested riparian zones (Murphy and Koski 1989, Carlson et al. 1990, Van Sickle and Gregory 1990, Beechie et al. 2001, Wing and Skaugset 2002) and its function in channels (e.g., Bilby and Ward 1991, Montgomery et al. 1995, Abbe and Montgomery 1996, Beechie and Sibley 1997, Collins et al. 2002), which are among the most studied of riparian functions. However, a similar assessment approach can be applied to other types of riparian systems and for different riparian functions.

As with watershed-level processes, there are two types of assessments required for reach-level processes. The first assessment identifies where reach level processes have been disrupted. The distribution of riparian conditions at this larger spatial scale can provide a general sense of the change in riparian function from historic conditions (e.g., Lunetta et al. 1997, OWEB 1999a). Subwatersheds where the current distribution of riparian conditions deviates markedly from that expected under a natural disturbance regime are locations where riparian restoration efforts may be appropriate. The same data can also help managers understand how different land use practices differ in their degree of impact on riparian functions. These relationships can then help assess the potential impacts of large-scale land use policies on salmon habitat recovery (e.g., evaluating potential effects of growth management legislation).

Because of limitations in the satellite classification of riparian forests, field inventories of riparian sites must be used to identify specific restoration actions (Clary and Leninger 2000, see our Appendix C). Field inventories may consist of initial measurements and classification from

aerial photography, combined with field confirmation of the riparian vegetation conditions for each stream reach. Newly developed multispectral technologies may also be of use in identifying riparian conditions with sufficient detail for site-scale planning purposes. At a minimum, they should classify riparian conditions by buffer width, stand type, and age of vegetation. From the data, managers can identify impaired or moderately impaired stream segments in order to determine the likely cause of that impairment, and identify required restoration actions. In general, impairment is defined with respect to a natural reference condition for the area in question, which is usually based on historical information (e.g., OWEB 1999a, Collins and Montgomery 2001, our Appendix C). However, in grazed riparian areas where the natural riparian vegetation is not forested, identification of disrupted riparian function may rely on measures of stubble height as an indicator of the level of disturbance (Clary and Leninger 2000, Turner and Clary 2001).

Regardless of current condition of riparian areas, establishing protected areas along the channel where natural riparian vegetation can develop through time and interact with the stream is a necessary component of riparian restoration. Active restoration efforts may be appropriate at currently impaired sites. Riparian restoration may include exclusion of livestock in drier riparian systems with less woody vegetation (Clary et al. 1996, Clary 1999), as well as the planting of desired riparian plant species or manipulation of the existing vegetation to accelerate tree growth and the development of desired stand structural characteristics (Berg et al. 1996, Beechie et al. 2000). Inventories of disrupted riparian functions in drier ecoregions are conceptually similar, but historical and present-day disrupted riparian communities will differ (e.g., the use of ecoregion information in OWEB 1999a).

Channel-floodplain interactions—Disruptions to floodplain and channel interactions may also dramatically reduce abundance of wood, pools, and off-channel habitats in larger river systems. These disruptions may result from altered sediment and wood supplies (e.g., downstream of dams) (Pohl 1999, Shafroth 1999), installation of dikes and riprap to control channel movement (e.g., Beechie et al. 1994, Collins et al. 2002), and channel incision that isolates a channel from its floodplain (e.g., Peacock 1994). Inventories that can help identify these disruptions include measurement of channels no longer accessible to salmon (Sedell and Luchessa 1982, Beechie et al. 1994), mapping of dikes, riprap, and disconnected floodplain surfaces (Appendix C), aerial photograph inventories of channel and habitat changes downstream of dams (Pohl 1999, Shafroth 1999), and inventories of incised channel segments.

Stream channel classification systems are another inventory tool that can be used to help identify disruptions in channel-floodplain interactions. Stream classification systems use the same suite of key variables such as geology, valley floor constraint, and channel slope to determine stream channel type and response of channels to changes in inputs such as water, wood, sediment, and energy over an entire watershed (Table 8). Stream channel classification systems reduces the number of variables and measurements needed to differentiate site response to channel-floodplain disruption, allows for the classification of spatial variability by grouping channels across a watershed, and creates a consistent method that can be used on a larger-scale across watersheds. The features used to classify channel reaches and valley segments by identifying common characteristics or patterns are often relevant for the development of channel-floodplain restoration plans, as stream reaches that have similar physical characteristics will respond to restoration actions similarly. These classification systems should be used in conjunction with the preceding inventory methods.

Habitat connectivity—impaired fish passage

Stream crossing structures that block fish access to useable habitats has been a recognized problem for many years in the Northwest. Such blockages can account for as much as 50% of lost smolt production from tributaries in Puget Sound river basins (Beechie et al. 1994, Beechie et al. 2001). Since 1992, the Washington State Department of Transportation has examined 3,216 fish crossings on state roads and documented 1,556 fish barriers (Johnson et al. 2001). Forest landowners, state agencies, and federal agencies in the state of Oregon have identified almost 3,000 culverts on fish bearing streams since 1997 (OPSW 2001). Private landowners, local, state, and federal agencies, and watershed groups have recognized magnitude of this problem for the last decade and have developed systematic methods to fix the problem through barrier inventory, assessment, and allocation of funds to correct the fish passage problems identified (ODFW 2001, WDFW 2001). For example, since 1995 over 900 locations throughout Oregon have been made more accessible to fish (OPSW 2001). Because there is not one comprehensive inventory database for all fish blockages in each state or throughout the Northwest, it is difficult to document the entire picture of what has been accomplished to date or the overall benefit to fish populations (OPSW 2001). Nevertheless this is an important component of restoration that has been ongoing and is needed for the recovery of Pacific Northwest salmon populations.

Assessing such isolation of habitats is one of the simplest inventories that can be conducted because criteria for fish migration blockages are relatively clear and identifying the amount of habitat affected involves little subjectivity. All the Northwest states have developed fish passage criteria for juvenile and adult salmonids that can be used as the basis for identifying fish blockages (ODFW 2001, WDFW 2001). Moreover, combining these inventory results with cost estimates for restoration actions allows resource managers to rank the cost-effectiveness of

individual projects in order to more effectively direct the expenditure of limited restoration funds.

For example, Snohomish County Public Works, Surface Water Management, in Washington, combined eight inventories that identify isolated habitat in the Stillaguamish River Basin from a variety of sources including Snohomish County Public Works, Washington Department of Fish and Wildlife, the Washington State Department of Natural Resources, the Stillaguamish Tribe, and the U.S. Forest Service (Footnote: M. Purser, Snohomish County Public Works, Surface Water Management, 2731 Wetmore Ave., Third Floor, Everett, WA 98201. Pers. commun., Sept. 2002) (Figure 14). Three out of the five agencies doing the inventories followed the Fish Passage Barrier and Prioritization Manual of Washington Department of Fish and Wildlife (WDFW 1998). Based on an inventory of 952 structures as of 2002, they identified 544 structures as 100% passable to salmonids, 337 structures as less than 100% passable, and 71 structures that as unknown due to no information (Footnote above: Mike Purser, SCPW, SWM, personal communication).

One of the inventories has been used to evaluate the cost-effectiveness of reconnection projects based on the habitat area upstream of the project, multiplied by the average life span of a blockage (~50 years) and divided by the cost of the project (Pess et al. in press). These results allowed natural resource agencies to identify the most cost effective projects for reconnecting blocked tributary habitats based on benefits to multiple salmonid species, as well as costs of reconstructing individual stream crossings (Pess et al. in press).

Water diversions can also impair fish passage, and have been a recognized problem for salmonids in the Northwest. There are almost 76,000 permitted water diversions in Oregon alone, however, many of these do not affect ESA-listed salmonids (OPSW 2001). Less than

1,000 of these diversions are required to be screened because they are greater than 30 cubic feet per second, however only five have been screened to date (OPSW 2001). NOAA Fisheries, California, Idaho, Oregon, and Washington have all developed criteria to exclude both juvenile and adult salmonids from being entrained in water diverted without being impinged on the diversion screens (NMFS 1995, CDFG 2000, ODFW 2001, WDFW 2001). The criteria developed by federal and state agencies allows for the development of inventories because it involves less subjectivity. Again, combining these inventory results with cost estimates for restoration actions allows managers to rank the cost-effectiveness of individual projects in order to more effectively direct the expenditure of limited restoration funds.

Assessing Biological Integrity

A key component of salmon habitat is the stream biota itself. Invertebrates, amphibians, diatoms, and other stream organisms are integral parts of the aquatic food web upon which threatened and endangered fish species depend. These assemblages are also sensitive to a variety of watershed disturbances expressed over multiple spatial scales, and therefore excellent indicators of stream condition. Unlike anadromous fishes that are subject to varied disturbances in both the marine and freshwater environment (e.g., migration blockages, interaction with hatchery fish, damaged estuarine habitats, or overharvest) less migratory stream organisms often provide a more accurate reflection of site condition. In particular, much research in the field of biological assessment (measuring and evaluating biota directly) has focused upon benthic invertebrates as indicator organisms (Rosenburg and Resh 1993, Merritt and Cummins 1996). Over the past century, bioassessment techniques using invertebrates and other assemblages have ranged from saprobien indexes (Hilsenhoff 1982), to toxicity testing (Buikema and Voshell 1993), indicator species abundance (Farwell et al. 1999), diversity indexes (Wilhm and Dorris

1966), and more recently to multivariate models (Wright et al. 2000) and multimetric indexes (Davis and Simon 1995).

In the context of ecosystem recovery planning, these bioassessment tools can be used to identify high quality areas in need of protection, degraded reaches in need of restoration, and assist in identifying the specific stressors causing biological impairment (such as factors discussed above in the preceding subsections of this section). This type of information will help focus assessments of disrupted ecosystem processes on those impacts that are most biologically important (Beechie and Bolton 1999). Of late, many studies incorporating specific bioassessment tools have evaluated the relationships between these measures of instream biological condition and land uses/land cover patterns over multiple spatial scales (Steedman 1988, Richards et al. 1996, Allan et al. 1997, Wang et al. 1997, Morley and Karr 2002). Some applications of this research include linking specific land use impacts and current condition of stream reaches, and setting realistic recovery goals given current land use patterns in particular river basins. We anticipate that these biological assessments will be most valuable in urban and agricultural areas where multiple impacts are likely to have occurred, and where the array of necessary process assessments may be prohibitively expensive without information to help prioritize them.

Multivariate models

In this approach a predictive model is developed based on a large (≈ 200 sites) data set of reference (minimally disturbed) sites (Reynoldson et al. 2001). Using multivariate statistical analyses, reference sites are matched to a set of habitat descriptors (e.g., stream order, elevation, etc.) and classified into groups. Level of impairment at a given sample site is then determined by comparison to the appropriate reference group. This approach has been most widely applied

with the development of RIVPACS (River Invertebrate Prediction and Classification System) in England, with AUSRIVAS (Australian River Assessment Scheme) in Australia, and with BEAST (Benthic Assessment of Sediment) in Canada (Wright et al. 2000). In the Pacific Northwest, multivariate models have been developed for the Fraser River basin in British Columbia (Reynoldson et al. 2001) and with the BORIS (Benthic Evaluation of Oregon Rivers) model in Oregon (Canale 1999). Based on benthic invertebrates, BORIS scores a site from 0 (severe impairment) to 100 (comparable to reference condition). A RIVPACS-type predictive model applicable to wadeable streams throughout Oregon, Washington, and Idaho is currently being developed (Hawkins and Ostermiller 2001, [Footnote] G. Hayslip, U.S. EPA, Region 10, 1200 Sixth Ave., Seattle, WA 98101. Pers. commun., March 2002).

Multimetric indexes

Multimetric indexes, such as an index of biological integrity (IBI), integrate empirically tested attributes (metrics) of stream biota (Karr and Chu 1999). This approach was first developed using fish communities in the midwestern United States (Karr et al. 1986), but has since been modified for a variety of assemblages—most commonly fish (Simon 1998), invertebrates (Kerans and Karr 1994), and algae (Hill et al. 2000). As with multivariate models, IBIs and other multimetric indexes are regionally calibrated based on ecoregion designations and local reference conditions. In the Pacific Northwest, an IBI using benthic macroinvertebrates was developed and calibrated with data from both Oregon and Washington (Kleindl 1995, Fore et al. 1996, Morley 2000, Adams 2001). This index, known as the benthic index of biological integrity or B-IBI (Karr and Chu 1999), is composed of ten measures of taxa richness, population structure, disturbance tolerance, and feeding ecology (Table 9). When scores from these metrics are summed, B-IBI provides a numeric synthesis of site condition that ranges from 10 (poor) to 50 (excellent), and can determine five categories of resource condition (Doberstein et al. 2000).

Multimetric indexes developed in other states differ somewhat in field methods and metrics (Hayslip in press).

Applications

While many of the current bioassessment protocols in use today across the nation were developed largely in response to legal mandates under the CWA, these monitoring tools have much application to recovery planning under the ESA. The CWA has as its main objective “to restore and maintain the chemical, physical, and biological integrity of this Nation’s waters” (PL 92-500, CWA, §101(a)). In response to this legal mandate, and under EPA guidance (Plafkin 1989, Barbour et al. 1999), states developed assessment protocols and water quality standards to determine if their water bodies are supporting beneficial uses such as recreation, domestic water supply, and—most pertinent to recovery planning efforts—aquatic life attainment. Although these assessment protocols and water quality standards were traditionally focused primarily on physical and chemical criteria, over the last decade there has been increasing incorporation of biological indicators which directly measure aquatic life and the maintenance of “biological integrity.”

Biological integrity has been defined in many ways; we use it here as defined by Karr (1991): “the capability of supporting and maintaining a balanced, integrated, adaptive community of organisms having a species composition, diversity and functional organization comparable to that of the natural habitat of the regions.” States and other public and private entities use information collected under the CWA reporting requirements in much the same way that it could be applied to ESA recovery planning; for example, in watershed assessments that inventory biological condition across large areas and quantify level of impairment; as a screening tool for identifying areas in need of further biological, physical, or chemical evaluation; in risk assessments, pollution permitting, and evaluation of proposed habitat modifications; and in

prioritizing areas in need of protection and restoration, and subsequently evaluating these conservation actions (Yoder and Rankin 1998, Karr and Chu 1999, Morley and Karr 2002). In the following paragraphs, we describe biological assessment protocols currently in development or in place throughout the Pacific Northwest.

In the Western United States, both multimetric and multivariate techniques are applied by state environmental agencies to assess and report on the biological integrity of surface waters. In Idaho, the Department of Environmental Quality has developed an ecological assessment framework for both wadeable streams and larger rivers composed of four multimetric indexes based on invertebrates, fish, diatoms, and physiochemical parameters (Grafe 2000a, 2000b). The California Department of Fish and Game uses a 19-metric invertebrate stream bioassessment procedure modified from the EPA's national Rapid Bioassessment Protocols (Barbour et al. 1999). The University of Alaska at Anchorage, working in conjunction with the state's Department of Environmental Conservation, is developing an invertebrate multimetric stream condition index for three stream types defined by gradient and substrate (Major et al. 2001). The Washington Department of Ecology and Oregon Department of Environmental Quality both currently apply a combination of multimetric and multivariate approaches to assess the condition of streams and rivers (Mochan and Mrazik 2000, Plotnikoff and Wiseman 2001). All of the states discussed above include narrative biological criteria in their water quality standards; Oregon is taking this a step further and will soon be incorporating numeric biocriteria in its standards (Footnote: G. Hayslip, U.S. EPA Region 10, pers. comm., 2002).

Although the state environmental agencies are largely responsible for defining, evaluating, and protecting designated uses of water bodies under the CWA, federal, tribal, and local regulatory agencies, volunteer, nonprofit, and private organizations, and academic

institutions also conduct bioassessments to varying degrees. At a regional level, monitoring work conducted under the Northwest Forest Plan includes bioassessment of invertebrates, periphyton, and fish (Reeves et al. 2002). In the Puget Sound region, B-IBI has been applied by city and county agencies (King County 1996, Thornburgh and Williams 2000), university scientists (May et al. 1997, Larson et al. 2001, Morley and Karr 2002), and volunteers (Fore et al. 2001) to track the health of streams over time, to screen watersheds for further physical or chemical monitoring, and to evaluate various restoration and conservation strategies. For example, two studies conducted recently in Washington State used B-IBI and other invertebrate metrics to evaluate the biological effectiveness of wood placement in forested (O’Neal et al. 1999) and urban basins (Larson et al. 2001).

Tuning restoration efforts to site-specific needs is enhanced by using biology to aid in the detection of the primary causes of degradation. Both multimetric indexes and multivariate analyses provide a numeric synthesis of the biological dimensions of site condition, but they can also be broken down to derive descriptive and potentially diagnostic information from each of the component metrics. For instance, in the case of aquatic invertebrates, there are hundreds of species throughout the western states—each with specific life history requirements and varying tolerance to specific forms of disturbance (Rosenberg and Resh 1993, Merritt and Cummins 1996). The groundwork has already been laid for research on biological response signatures: “biological community characteristics that aid in distinguishing one impact type over another” (Yoder 1991, Yoder and Rankin 1995); what remains for future research is to better link biological response variables with physical and chemical manifestations of human disturbance. Nor is bioassessment limited to lotic waters; protocols are currently being developed for lake (Gerritsen et al. 1998), wetland (Adamus et al. 2001), and estuarine environments (Gibson et al.

2000). This evolving body of work will enable resource managers to better track ecological health over a larger portion of the salmon landscape.

Conclusions

Assessments of the current and historical conditions of a watershed can greatly improve our efforts to plan, implement, and monitor ecosystem restoration for the recovery of Pacific salmon. Systematically collected habitat data, a more thorough understanding of fish responses to habitat change, and a greater understanding of stream biota will allow refinement of the modeling tools used to predict fish and other biological response from application of different restoration strategies. These refinements will improve estimates of rates and pathways of recovery for many salmonid species in any river, and assist in prioritizing restoration actions. However, many of these refinements are still several years from completion.

In the interim, systematic inventories of disrupted habitat-forming processes and blockages to salmon migration should be conducted to provide a complete river basin overview of necessary restoration actions that can be prioritized and sequenced logically. A minimum set of inventories for any river basin should include barrier inventories, landslide inventories, floodplain and riparian characterization, channel and valley type classification, and road and biological indicator inventories. Some of these data are already available for parts of many watersheds. These data provide the basis for identifying needed restoration actions, which can be prioritized by cost-effectiveness, influence on particular species, adjacency to existing population centers or centers of biological diversity (commonly called refugia, biological hot spots, source watersheds, core areas, key habitat), or other strategies.

There are many sources of uncertainty in these assessments. Uncertainties in assessments stem from natural variability in habitat-forming processes, habitat characteristics, and fish

populations, as well as from errors in assumptions and limitations of data or knowledge. Our ability to characterize these types of uncertainty is limited by availability of data on watershed processes, habitat conditions, and fish populations over long periods of time. Lack of knowledge about current habitat conditions or responses of fish populations to changing habitat conditions introduce uncertainty into predictions of fish responses to watershed and habitat restoration. Improving the quality of the data reduces uncertainty related to knowledge gaps and improves ability to address the uncertainty related to natural variability in fish response to habitat conditions.

Recovery plans designed to protect and recover processes that create and sustain riverine habitats are more likely to recover salmon of all species. Use of a comprehensive assessment process and developing restoration plans focused on the reestablishment of habitat-forming processes minimizes conflicts that can arise with species-centric restoration approaches. Restoration of habitat-forming processes targets restoration of the natural array of habitat types and conditions within a watershed, which is consistent with the concepts of watershed and ecosystem management supported by the scientific community. Moreover this approach focuses on the natural potential of each watershed, and therefore is most likely to restore the diversity and abundance of stocks appropriate to each watershed in Puget Sound.

Table 7. Examples of methods for rating individual landscape processes.

Distributed watershed processes

Hydrology—disruption of the peak flows, low flows, and channel forming flows

A change in the magnitude, frequency, duration, and rate of minimum, mean and maximum flows can be examined using the indicators of hydrologic alteration (IHA), which assess the difference in 32 biologically significant hydrologic parameters of pre- and post-anthropogenic activities such as dam operations, flow diversion, groundwater pumping, or intensive land use (Ritcher et al. 1997).

Hydrology—increases in peak flows

Lowland basins: Hydrologic impairment in lowland basins can be rated based on planned effective impervious area (EIA), which is the weighted average EIA upstream of the stream reach under fully developed conditions. $EIA \leq 3\%$ is considered “functioning”, EIA between 3% and 10% is “moderately impaired,” and $EIA > 10\%$ is “impaired” (based on Booth and Jackson 1997, and see example in our Appendix C).

Mountain basins: Peak flow ratings for mountain sub-basins can be developed based on empirical correlations between land use and elevated peak flow in forested basins (Jones and Grant 1996, and see example in our Appendix C).

Sediment supply—an increase in sediment supply

Estimating impairment of sediment supply: Changes in average sediment supply for forested sub-basins within a watershed can be estimated based on present-day sediment supply rates from unlogged, clear-cut, and roaded portions of the watershed (Dietrich and Dunne 1978, Paulson 1997, Montgomery et al. 1998).

Surface erosion on agricultural and rangelands: Changes in average sediment supply for croplands within a watershed can be estimated on crop practices, soil type, rainfall, slope, and other factors (Dunne and Leopold 1978).

Inventory—identify sediment reduction projects: Inventories must focus on factors that influence sediment supply, identification of landslide hazard areas so that forest practices can be avoided or modified in sensitive areas (e.g., Montgomery et al. 1998), such as risk of road-related landslides (e.g., Renison 1998), crop management practices that increase surface erosion (Wischmeier and Smith 1965), or grazing practices that alter sediment supply.

Sediment supply—a decrease in sediment supply

Routing estimates: In-channel sediment storage budget (Madej and Ozaki 1996)

Disrupted water quality

Contaminants

Association of fish assemblage structure and environmental variables: Compare fish assemblage composition to chemical and physical environment. (Waite and Carpenter 2000).

Table 7 continued. Examples of methods for rating individual landscape processes.

Reach-level processes

Riparian function

Remote sensing assessment: Use remote sensing classifications of vegetation to assess riparian buffer width and type in order to help determine how much and where riparian buffer impairment has occurred on a reach, watershed, and river basin scale. Identify historic conditions using reference locations and historic documentation. Compare historic condition to current riparian condition in order to determine degree of change.

Field inventory: In addition to documenting forested buffer width, field inventories also classify stand types by species mix and seral stage, which gives sufficient information to prescribe generalized management regimes for each segment of riparian forest. Inventories also identify areas of livestock access and potential fencing projects.

Riparian alteration due to grazing: Use similar methods but include indicators such as stubble height measurements as indicator of disturbance (Clary and Lenninger 2000, Turner and Clary 2001).

Channel and floodplain interactions

Floodplain areas can be delineated using 100-year floodplain maps from the Federal Emergency Management Agency maps or U. S. Geological Survey 7.5-minute quadrangles and aerial photographs.

Habitat connectivity—anthropogenic blockages

Man-made barriers to anadromous fish habitat are identified through a systematic field inventory of channel crossing structures (culverts, tide gates, bridges, dams, and other man-made structures). An inventory identifies the type and physical dimensions of structures as well as physical attributes necessary for modeling water flow conditions and comparing results to passage criteria for salmonids (e.g., WDFW 1998).

Table 8. Summary of contemporary spatial scale classifications (adapted from Bauer and Ralph 1999).

Classification system reference	Spatial scales addressed by classification system						
	Eco- region	River Basin	Watershed	Sub- watershed	Valley segment	Stream reach	Habitat unit
Bisson et al. 1982							X
Frissell et al. 1986			X	X	X	X	X
Seaber et al. 1987		X	X	X			
Paustian 1992				X	X	X	
Maxwell et al. 1995	X	X	X	X	X	X	
Rosgen 1994					X	X	
Montgomery and Buffington 1997						X	
Omernik and Bailey 1997	X	X	X				

Table 9. The 10 metrics of the B-IBI and their predicted response to increasing human disturbance.

Metric	Description	Response
<i>Taxa richness and composition</i>		
Total taxa	Richness	Decrease
Mayfly taxa	Richness	Decrease
Stonefly taxa	Richness	Decrease
Caddisfly taxa	Richness	Decrease
<i>Population structure</i>		
Dominance by top 3 taxa	Relative abundance	Increase
Long-lived taxa richness	Richness	Decrease
<i>Tolerance and intolerance</i>		
Intolerant taxa richness	Richness	Decrease
Tolerant taxa	Relative abundance	Increase
<i>Feeding and other habits</i>		
Clinger taxa richness	Richness	Decrease
Predators	Relative abundance	Decrease

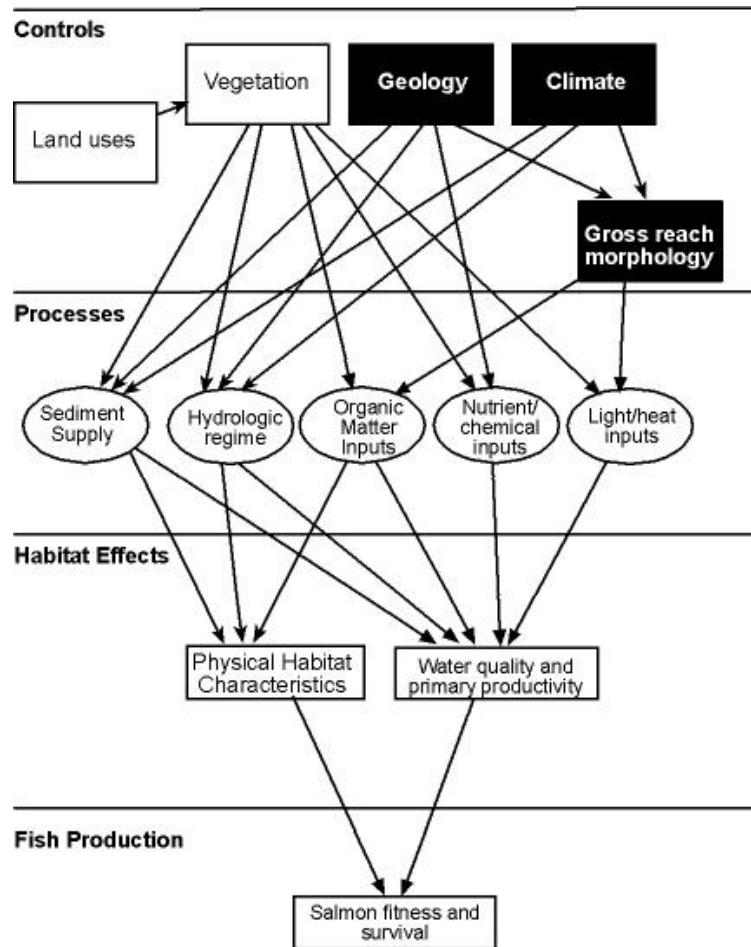


Figure 11. Schematic diagram of relationships between controls on watershed processes, effects on habitat conditions, and salmon survival and fitness (adapted from Beechie and Bolton 1999). Dark boxes in upper row are ultimate controls, light boxes are proximate controls.

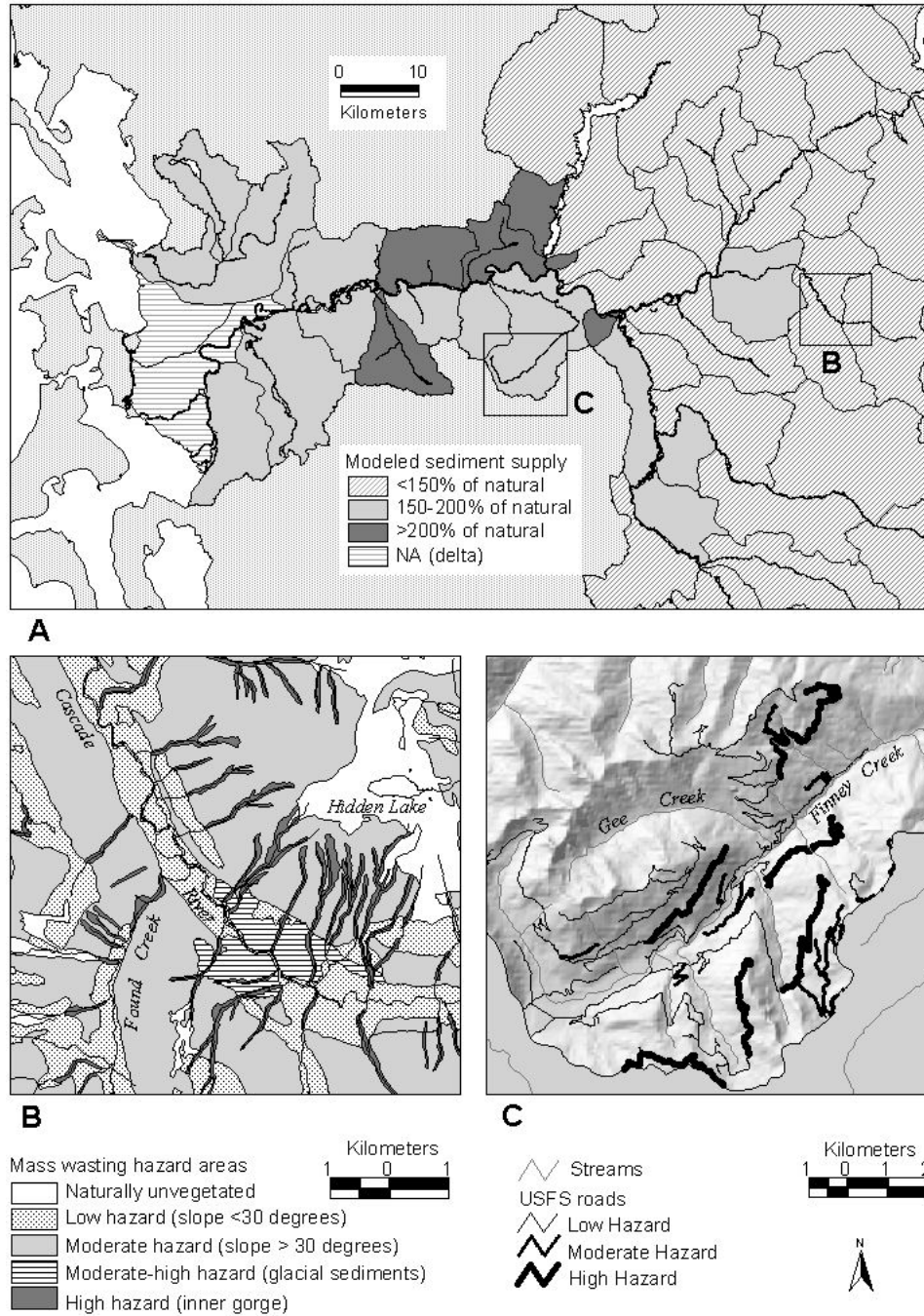


Figure 12. (A) is map of areas in the Skagit River Basin where sediment supply has likely increased due to land use, based on extrapolation of data from sediment budgets. (B) is landslide hazard map for a portion of the upper Cascade River Basin. (C) is hazard map of U.S. Forest Service roads classified as high risk of failure, moderate risk, or low risk.

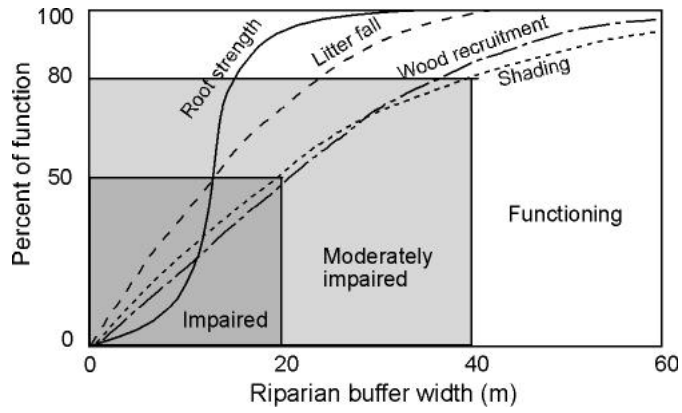


Figure 13. Illustration of change in riparian function with distance from channel (curves adapted from Sedell et al. 1997), and the Skagit Watershed Council's classification of impaired, moderately impaired, and functioning riparian forests.

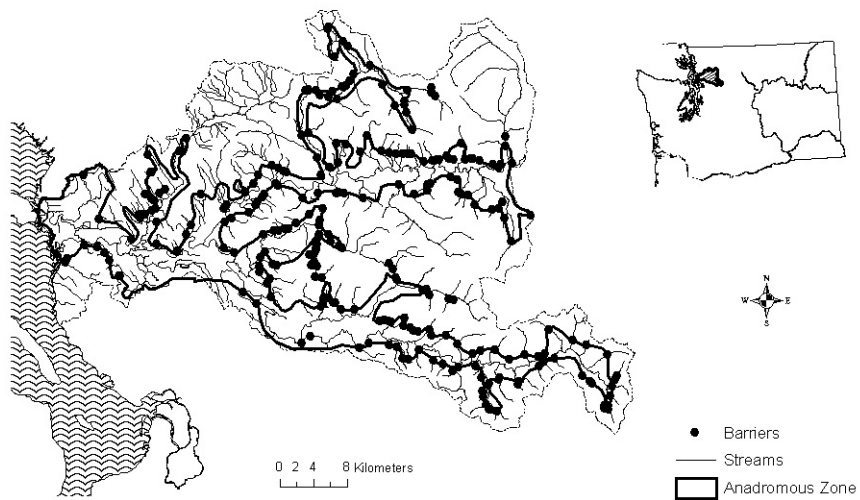


Figure 14. Example map of a portion of the inventoried stream crossing structures for a portion of the Stillaguamish River Basin. The map shows structures that are blocking. Dark line indicates the historical extent of anadromous fish.

PRIORITIZING POTENTIAL RESTORATION ACTIONS WITHIN WATERSHEDS

Previous sections outlined habitat analyses for setting recovery goals (Phase I planning) and identifying ecosystem recovery actions (Phase II planning). Following these assessments, the next step is to incorporate Phase I and Phase II information into a recovery plan and develop a prioritized list of ecosystem restoration actions within watersheds (Beechie and Bolton 1999, OWEB 2001, Roni et al. 2002). If models could accurately link changes in watershed processes to habitat change and population responses, this would be a simple matter of running the models to see which restoration actions would most efficiently attain the recovery goals. However, such models do not currently exist, and we must rely on a longer term approach to recovery that develops initial hypotheses about which actions will be most effective, conducts management experiments to test these hypotheses, and monitors effectiveness of different actions to adjust the recovery plan in the future. In this section we describe a range of prioritization approaches that can be used in recovery planning, as well as the need for management experiments and monitoring.

What Is Restoration and What Do We Know about It?

The term “restoration” has been used to describe a suite of stream, watershed, and estuarine habitat manipulations, enhancements, and improvements. As we said in the Introduction section of this technical memorandum, restoration in its strictest definition is returning a site to some predisturbance condition (Gore 1985, NRC 1996). Generally it is more holistic or systemic than habitat creation, reclamation, rehabilitation, or enhancement, and not accomplished through manipulation of individual ecosystem or watershed elements (NRC 1996,

Frissell and Ralph 1998). In contrast, habitat enhancement is the improvement of habitat from its existing or previous condition. It does not necessarily seek to restore conditions to some predisturbance state or seek to restore disrupted watershed or ecosystem processes and functions such as delivery of water, wood (organic material) and sediment. Restoration can also be further classified as passive and active (Kauffman et al. 1997). Passive techniques seek to restore processes by halting detrimental land uses, protecting areas and setting up conditions that will allow recovery of the stream (e.g., exclusion of cattle from riparian areas, protecting a riparian area). Active restoration generally seeks to create relatively rapid habitat changes (within a few months or years) in watershed processes or habitat conditions. Active techniques, which may include habitat enhancement, are those that seek to directly improve or manipulate watershed processes or habitat, such as thinning of riparian areas, or removal of migration barrier or placement of logs in a stream channel to create pools. Here we use the term restoration generically to mean both restoration and enhancement, but we distinguish between those activities that restore watershed or ecosystem processes and those that enhance habitat (Figure 15).

Before one can prioritize specific restoration actions within a watershed, one needs a comprehensive understanding of the physical and biological effectiveness of various restoration methods. Most restoration techniques fall into five general categories: 1) habitat reconnection, 2) road improvement, 3) riparian restoration, 4) instream habitat restoration, and 5) nutrient enrichment. Within each of these five general categories, several specific techniques can be identified. For example, several types of wood placement have been developed to increase instream habitat complexity and loss of large woody debris (LWD).

Gore (1985), Reeves et al. (1991), Slaney and Zaldakas (1997), Cowx and Welcome (1998), OWEB (1999b), and others provide descriptions of various restoration techniques and information on design and implementation of these techniques. Though rather limited information exists for most restoration techniques, several authors have reviewed the effectiveness of different techniques (Gore 1985, Reeves et al. 1991, Roni et al. 2002). Roni et al. (2002) reviewed common restoration techniques, their effectiveness, longevity and whether they restore processes or are short-term habitat enhancement (Table 10). Although many authors have discussed the need to restore processes and prioritize restoration (Beechie et al. 1996, Minns 1996, Jones and Moore 2000, Rieman et al. 2000, Luce et al. 2001, JNRC 2002), specific guidance on which techniques to use and how to prioritize restoration is rather limited. In part, the lack of guidance on the appropriateness of various techniques and how to prioritize restoration actions stems from limited information on the biological effectiveness of various restoration and enhancement techniques (Reeves et al. 1991, Chapman 1996, Roni et al. 2002, 2003). The responses of fishes to watershed and stream habitat restoration techniques have not been thoroughly evaluated, and there is considerable debate within the scientific community about the effectiveness of various techniques (Reeves et al. 1991, Kondolf 1995, Kauffman et al. 1997, Roni et al. 2002).

Most monitoring has focused on the physical response to various instream restoration techniques with inadequate monitoring of fish, invertebrates, and other biota. However, the biological response to various restoration techniques is often the ultimate measure of effectiveness. Monitoring of juvenile and adult salmonid abundance often requires more than 10 years to detect a response to restoration due to high interannual variability (Bisson et al. 1997, Reeves et al. 1997, Maxell 1999, Ham and Pearsons 2000, Roni et al. 2003). Moreover, some

techniques such as wood and boulder placement in streams yield highly variable results (Chapman 1996), and how the results of reach-level studies can be interpreted for population level responses remains unclear. Therefore, drawing conclusions about the biological effectiveness of various techniques has been difficult and that difficulty has hampered efforts to provide scientific guidance on restoration activities.

In the 1990s the notion that restoring watershed processes is the key to restoring watershed health and improving fish habitat throughout western North America and elsewhere became widely accepted. Beechie et al. (1996), Kauffman et al. (1997), Beechie and Bolton (1999), Roni et al. (2002) and others have described restoration and recovery strategies that place emphasis on restoring physical and biological processes that create healthy watersheds and high-quality habitats. Yet activities that restore processes (e.g., road removal and restoration, culvert removal, and riparian and upslope restoration) are often conducted at the site or reach level. Prioritization of restoration actions needs to place site-specific restoration within a watershed context.

Strategies for Prioritizing Actions

Watershed and stream restoration are key components of many land management plans, and should be an important component of most recovery plans for threatened and endangered species. However, how individual site-specific actions might fit into a larger context of watershed restoration and salmon recovery is often unclear, and many approaches to prioritizing restoration actions are available. Deciding which approach best suits a specific watershed or population may depend on many factors, including how much is known about habitats that limit recovery of populations, how much is known about the causes of habitat degradation in a

watershed, or even the number of listed species within a watershed. Therefore, describing a single prioritization scheme that will be applicable to all watersheds is difficult.

In this section we describe several strategies that may be used to prioritize restoration actions, beginning with an interim approach to prioritizing site-specific restoration activities in a watershed context. This interim approach is based primarily on the effectiveness and longevity of specific types of restoration actions, and does not require detailed information about which habitats limit population recovery. We then describe a second approach for watersheds where only one species is listed and priorities may be directed toward those actions that are most likely to benefit the listed species. Finally, we compare several alternative approaches to illustrate how priorities might differ under different prioritization schemes. We focus on those activities that occur within an individual watershed (U.S.G.S. 5 or 6th field hydrologic units - HUCs). Other large-scale restoration efforts that may occur at the basin or ESU scale, and thus may influence many watersheds, should be prioritized within a basin or ESU. We do not discuss such broad-scale priorities here.

An Interim Approach

In the absence of detailed knowledge of factors limiting recovery, an interim prioritization scheme can serve to logically sequence different types of restoration actions based on their probability and variability of success, response time, and longevity (Table 10). All else being equal (e.g., costs, listed species concerns), those techniques that have a high probability of success, low variability among projects, and relatively quick response time should be implemented before other techniques. For example, reconnecting isolated off-channel habitats or blocked tributaries provides a quick biological response, is likely to last many decades and, based on available evidence, has a high likelihood of success. Generally, these types of

restoration activities should be undertaken before methods that produce less consistent results. Riparian restoration or road improvement may not produce results for many years or even decades for some functions (Table 10) and should be considered after reconnecting high-quality isolated habitats. Other techniques such as instream LWD placement or other instream restoration are generally effective at increasing coho salmon densities, but less certain for other species. Instream actions such as LWD placement are often habitat manipulations or enhancements and should be undertaken either after or in conjunction with reconnection of isolated habitats and other efforts to restore watershed processes. In addition, manipulation of instream habitat may be appropriate where short-term increases in fish production are needed for a threatened or endangered species (Beechie and Bolton 1999).

Using information summarized in Table 10, Roni et al. (2002) developed a hierarchical flow chart that can be used to help guide the selection and prioritization of restoration projects based on understanding of watershed processes and current knowledge on effectiveness of different techniques (Figure 16). This flow chart combines the known effectiveness of various techniques with the need to restore habitat-forming processes (Figure 15) identified by watershed assessments (the “Habitat Analyses for Phase II Recovery Planning: Identifying Ecosystem Restoration Actions” section, page XXX) and the protection of high-quality habitats. Protection of high-quality habitat should be given priority over habitat restoration, as it is far easier and more successful to maintain good habitat than to try and recreate or restore degraded habitat.

While most techniques fit well into this hierarchy, estuarine restoration, carcass placement and nutrient enhancement are relatively new techniques whose place in this hierarchy is uncertain. Little is known about the effectiveness of estuarine restoration, though reconnecting isolated estuarine habitats such as distributary sloughs is similar to reconnecting

isolated off-channel habitats, which has been shown to be effective (Table 10). Given the importance of estuaries to anadromous fishes and the success of reconnecting isolated off-channel habitats, it is likely that reconnecting estuarine habitat would be effective and should be considered at the same time as reconnecting other isolated habitats. The placement of salmon carcasses or other nutrients into streams may increase fish condition and production in the short term. This restoration technique is a form of habitat enhancement that can occur at any stage in the watershed restoration process. However, because it does not restore but rather mitigates for a deficient process, we have suggested that it be considered at the same point in the hierarchy as instream habitat manipulation. Similarly, the creation of new estuarine or off-channel habitats does not restore a process and the effectiveness of these efforts is unclear.

A common restoration technique not covered in Roni et al. (2002) is restoration of instream flows or natural hydrology either from water withdrawal projects or below large water storage projects. Water withdrawal or flow manipulation disrupts hydrologic processes, including delivery and routing of sediment and nutrients, and can dramatically impact habitat formation, connectivity, and quality (Bednarek 2001). We consider restoring instream flows and natural hydrologic patterns part of reconnecting isolated habitats, and therefore do not have a separate category for this technique. Upslope activities and land use can have dramatic effects on stream hydrology, sediment delivery, water chemistry, and water quality. Altering land use and other upslope restoration techniques were also not explicitly discussed in Roni et al. (2002) either, but essentially can be included with riparian restoration (Table 10).

Within the broad restoration categories in Figure 16, some techniques are more effective than others or more applicable in some provinces than others. For example, we include riparian silviculture with fencing and reduced grazing under riparian restoration. Livestock exclusion is a

form of riparian protection that has been shown to be effective on range and agricultural lands (Platts 1991), while the long-term effectiveness of riparian replanting and conversion in forested watersheds techniques is largely unknown. Priorities for different types of riparian restoration will differ by region and watershed, as will other specific restoration techniques that fall into the broad categories we have defined. However, a watershed assessment is the important first step to determine the most effective type of restoration within a given restoration category for the watershed in question (Beechie et al. in press).

The principles outlined above and in Figure 15 were designed primarily for forest, range, and other moderately modified rural lands. However, they are still useful in urban and agricultural lands, even though other factors such as large infrastructure (e.g., highways and buildings) may constrain certain restoration opportunities. In urban areas, hydrologic and sediment processes in streams are highly altered (e.g., increased high flows and channel down-cutting). Areas with intensive agriculture often have severe water quality problems, and stream channels in both urban and agricultural areas are often highly channelized and lack adequate riparian vegetation. Thus the framework we outline may need to be modified for use in these highly altered systems where some processes cannot be reliably restored, or where water quality or hydrologic changes may compromise the effectiveness of many of the commonly employed restoration techniques.

Single Species Approaches

In watersheds where most Phase I and Phase II analyses have been completed, a more detailed prioritization of actions is possible. To do this, one should know:

1. Which habitats are most impaired or most likely inhibiting recovery of populations (from Phase I assessments)?

2. Which impaired ecosystem processes prevent recovery of the limiting habitats (from Phase II assessments)?

The information from Phase I assessments can be used to identify high priority areas for salmon recovery. Because habitat use varies by species and life history pattern, the same list of habitat recovery areas might be prioritized differently depending on the species of interest (e.g., Beechie and Bolton 1999) (Figure 17). In general, habitat areas should be classified at a relatively coarse level of resolution (e.g., estuary, mainstem, overwintering habitats) because the information available for evaluating which habitats limit salmon recovery is very sparse, and the certainty of the answers is very low (see also the “Managing Uncertainty in Habitat Recovery Planning” section, page XXX). (We note that while EDT analyses have a finer spatial resolution, their results should be used only in the most general terms because of the large number of uncertainties inherent in 1) the collection of information, 2) the models used to estimate fish responses, and 3) the lack of model validation.)

Once specific habitat areas have been prioritized, the results of Phase II assessments can be examined to determine which restoration actions are required for recovery of the high priority habitat areas. The Phase II assessments identify the causes of habitat degradation by evaluating where different ecosystem processes have been disrupted and inventorying specific causes of habitat loss (e.g., barrier inventory, road sediment reduction inventory, riparian inventory). Thus, if Phase I assessments identify a specific habitat area as a likely limiting habitat (e.g., the estuary or tributaries for early rearing), then Phase II assessments identify the causes of degradation within that area (see also Appendix C, page XXX, for examples).

It is critical to bear in mind that the prioritization of actions does not alter the types of actions that are needed to restore ecosystems that support salmon. Rather, the Phase II

assessments identify the suite of actions needed to restore ecosystem processes and functions, and prioritization merely alters the sequence in which those actions are taken (Figure 17). This of course implies that restoration of ecosystems to support salmon will include a wide range of recovery actions affecting the entire life cycles of multiple species. However, where a single species is listed, altering the sequence of those actions for most rapid recovery of the listed species may be prudent.

Alternative Prioritization Schemes

Alternative strategies for prioritizing restoration that incorporate economics, biologic, ecologic, and biologic factors have been proposed or used, especially where there are multiple listed species and attempting to prioritize actions based on the needs of individual species will lead to conflicting priorities. In such cases other approaches to prioritization may be more appropriate, such as the refugia approach (Sedell et al. 1990) or a multispecies, cost-effectiveness approach (e.g., SWC 1998). Where at least one species appears to be at high risk of extinction, the refugia approach may be most appropriate to make sure that individual populations are preserved first. By contrast, watersheds with relatively stable populations might embark on a longer term, process-based approach to ecosystem recovery. It is likely that most recovery plans will incorporate different strategies for different watersheds or populations.

For example, Sedell et al. (1990), Wasserman et al. (1995), Beechie et al. (1996), Frissell (1993a), Frissell and Bayles (1996), and others have outlined restoration strategies that focus on providing refugia and protecting high quality habitats. Beechie et al. (1996) outlined a prioritization strategy that focused on providing refugia for a depressed steelhead trout stock in Deer Creek, Washington. Other strategies might prioritize actions on potential increase in fish numbers, cost, cost per fish, aquatic diversity, assuring for metapopulations structure or

diversity, or scoring based on a suite of these and other factors (e.g., Beechie et al. 1996, Frisell and Bayles 1996, Doyle 1997, SRSRC 2002, LCFRB 2002). Some states such as Oregon have developed sequential methodologies for conducting assessments, prioritizing, and implementing restoration activities (Figure 18). These various strategies incorporate management goals beyond simply restoring watershed or ecosystem processes and habitat. Thus the sequencing of restoration actions under different prioritization strategies will vary.

We demonstrate how priorities might differ based on different restoration prioritization schemes by running alternative scenarios. First, we developed a hypothetical list of potential restoration actions along with detailed information on their cost, length and area restored, whether they provide refugia for endangered species, potential increase in fish numbers, and cost per fish (Table 11). Second, we ranked restoration actions using different prioritization schemes mentioned above (Table 12). This analysis demonstrated that if restoration actions were prioritized based on Roni et al. (2002), impassible culverts and reconnection of habitats would occur first, followed by road, riparian, and LWD placement. If actions were prioritized by whether they were in a refugia for an ESA-listed species, instream flow and LWD placement would be first, simply because they are in a high priority area. Similarly, different cost, cost/fish, total fish production all produced slightly different prioritization scenarios. This simple example illustrates how priorities might differ based on the method, information used, and management objectives. In the next section, “Managing Uncertainty in Salmon Habitat Recovery Planning,” we also discuss accounting for uncertainty in prioritizing restoration actions toward incorporating the risks and likelihoods of success and failure (physical, biological, or financial) into the planning process.

The appropriate method for prioritizing restoration activities within a watershed will depend on numerous factors. Our intent in this section has been to discuss how one prioritizes site-specific restoration actions within a watershed or basin. However, all else being equal or if limited information is available, we would recommend a strategy similar to that outlined in Roni et al. (2002, see Figure 16) that focuses on reconnecting isolated habitats and restoring watershed processes before or alongside habitat manipulations or enhancement.

Need for Monitoring and Management Experiments

Reviews of various restoration techniques (e.g., Roni et al. 2002) indicate that knowledge about the effectiveness of most techniques is incomplete, and comprehensive research and monitoring are needed. Even techniques that appear to be well studied such as instream LWD placement need more thorough evaluation and long-term monitoring. This emphasizes the need for comprehensive monitoring and evaluation of both individual and multiple restoration actions at multiple scales. Many restoration actions should be treated as management experiments and accompanied by research and monitoring to determine both physical and biological responses. These results can then be used to guide future restoration actions and more accurately quantify the potential increase in fish production for habitat manipulations. Ultimately, thorough monitoring and evaluation of restoration actions will help us prioritize restoration opportunities and wisely spend limited restoration and recovery funds for salmon.

Table 10. Typical response time, duration, variability of success, and probability of success for common restoration techniques (modified from Roni et al. 2002).

Restoration type ^a	Specific action	Years to achieve response	Longevity of action (years)	Variability of success among projects	Probability of success
<i>Reconnect habitats</i>	Culverts	1-5	10-50+	Low	High
	Off channel	1-5	10-50+	Low	High
	Estuarine	5-20	10-50+	Moderate	Moderate to high
	Instream flows	1-5	10-50+	Low	High
<i>Roads and land use</i>	Road removal	5-20	Decades to centuries	Low	High
	Road alteration	5-20	Decades to centuries	Moderate	Moderate to high
	Change in land use	10+	Decades to centuries	Unknown	Unknown
<i>Riparian restoration</i>	Fencing	5-20	10-50+	Low	Moderate to high
	Riparian replanting	5-20	10-50+	Low	Moderate to high
	Rest-rotation or grazing strategy	5-20	10-50+	Moderate	Moderate
	Conifer conversion	10-100	Centuries	High	Low to moderate
<i>Instream habitat restoration</i>	Artificial log structures	1-5	5-20	High	Low to high ^b
	Natural LWD placement	1-5	5-20	High	Low to high ^b
	Artificial log jams	1-5	10-50+	Moderate	Low to high ^b
	Boulder placement	1-5	5-20	Moderate	Low to high ^b
	Gabions	1-5	10	Moderate	Low to high ^b
<i>Nutrient enrichment</i>	Carcass placement	1-5	Unknown	Low	Moderate to high
	Stream fertilization	1-5	Unknown	Moderate	Moderate to high
<i>Habitat creation</i>	Off channel	1-5	10-50+	High	Moderate
	Estuarine	5-10	10-50+	High	Low
	Instream	See various instream restoration techniques above			

^a The first three categories of restoration (reconnect isolated habitats, roads and land use, and riparian restoration) are considered process-based or passive restoration, the last three (instream, nutrient enrichment, and habitat creation) are considered enhancement or active restoration.

^b Depends on species and project design.

Table 11. Example of list of potential restoration actions within a watershed. All stream names and numbers are fictitious and for demonstration purposes only.

Site ID	Site name	Action	Refugia ^a	Km treated	M ² treated	Coho smolts ^b	Chinook smolts ^b	Cost	\$ per coho
A	Clark Creek	LWD placement	2	3	15,000	3,750	750	50,000	13.30
B	Upper Simpson Creek	LWD placement	1	2	10,000	2,500	500	32,000	12.80
C	Lower Simpson Creek	LWD placement	1	2	14,000	3,500	700	35,000	10.00
D	Check Creek	Fencing	1	5	60,000	6,000	3,000	20,000	3.30
E	Dry Creek	Increase flows	2	20	200,000	20,000	10,000	500,000	25.00
F	Big River	Reconnect tidal channels	3	1	100,000	10,000	50,000	350,000	35.00
G	Big River	Create new estuarine slough	3	2	200,000	20,000	100,000	750,000	37.50
H	Clark Creek	Culvert replacement/fish passage	2	3	15,000	3,750	0	150,000	40.00
I	Simpson Creek	Road de-commissioning	1	20	200,000	20,000	10,000	1,500,000	75.00
J	Clark Creek	Road resurfacing/sediment reduction	2	10	50,000	5,000	2500	750,000	150.00
K	Big River Slough	Reconnect isolated oxbow slough	3	4	800,000	400,000	40,000	75,000	0.19

^a Refugia numbers, based on Beechie et al. 1996, are: 1 = refugia (areas where recovery is relatively predictable); 2 = key habitat areas or areas that provide for the largest long-term recovery of species of interest, but are sensitive to disturbance and more difficult to restore; and 3 = key habitat areas or areas expected to provide the smallest gain for species of interest.

^b Numbers represent expected annual increase in smolt production.

Table 12. Example of different order of priorities based on use of different prioritization schemes using the fictitious information presented in Table 11. Roni et al. (2002) and refugia methods of prioritization do not distinguish between projects of the same type. Hence there are only four levels and three levels for the two methods, respectively.

Site ID	Potential restoration action	Roni	Refugia	Total cost	Cost/coho	Total fish
A	LWD placement	3	2	4	5	4
B	LWD placement	3	1	2	4	1
C	LWD placement	3	1	3	3	3
D	Fencing/cattle exclusion	3	1	1	2	6
E	Instream flows/purchase water rights	1	2	8	6	8
F	Reconnect estuarine tidal channel (dike removal)	1	3	7	7	9
G	Excavate new estuarine slough	4	3	10	8	10
H	Culvert replacement/fish passage	1	2	6	9	2
I	Road decommissioning	2	1	11	10	7
J	Road resurfacing/sediment reduction	2	2	9	11	5
K	Reconnect isolated oxbow slough	1	3	5	1	11

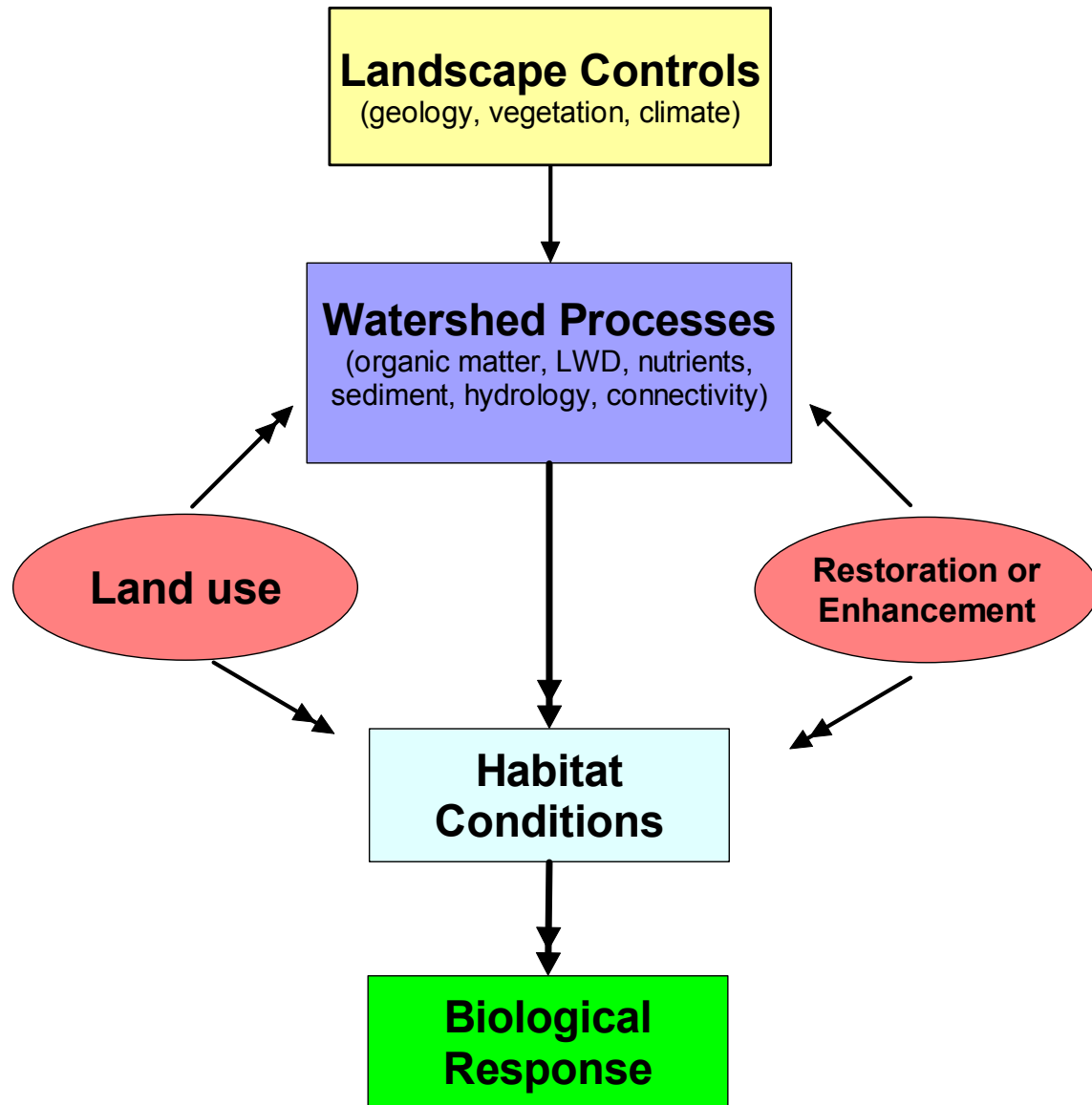


Figure 15. Simplified model of watershed controls, processes, and function and how land use, restoration and enhancement can influence habitat and biota.

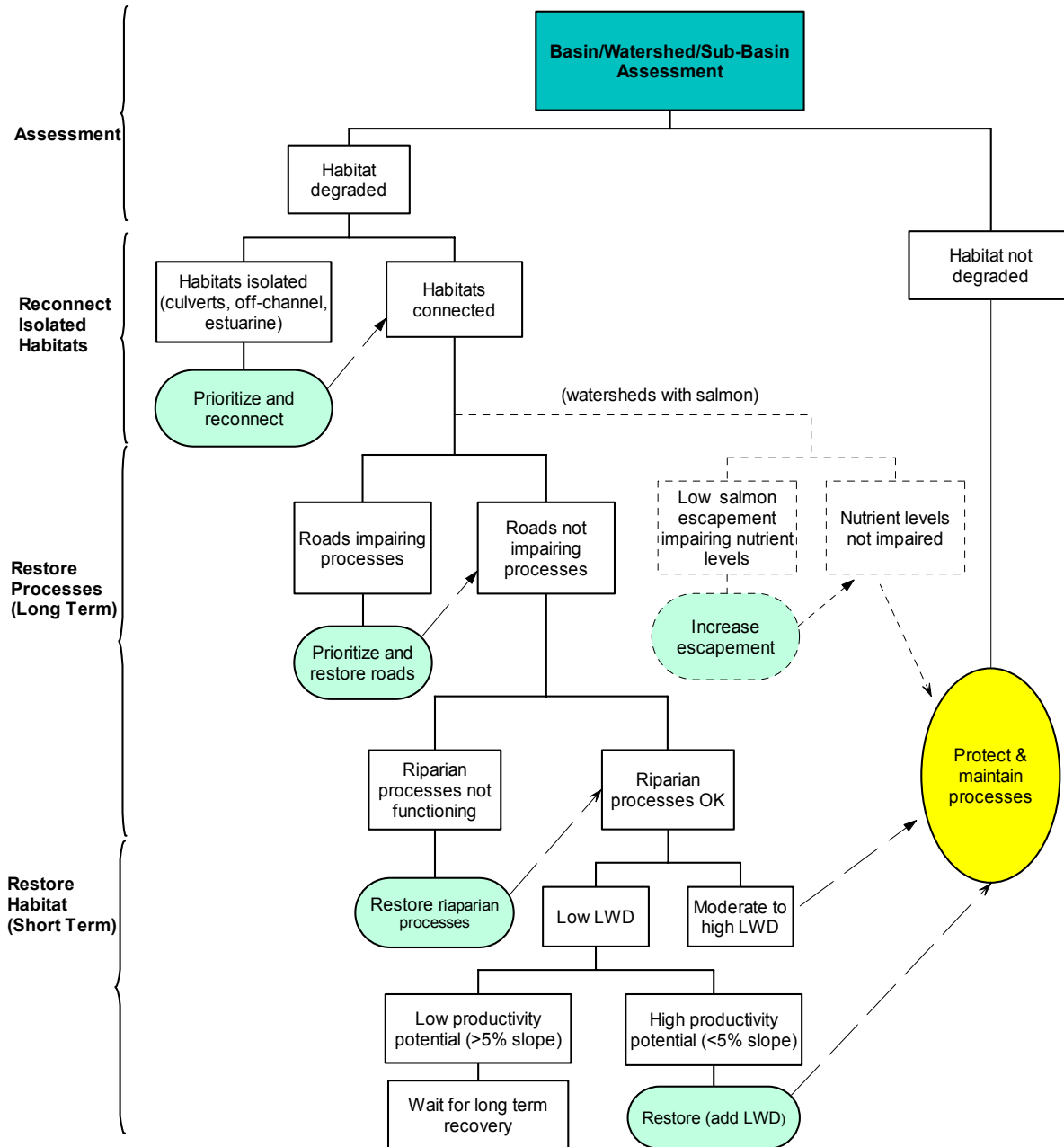


Figure 16. Flow chart depicting hierarchical strategy for prioritizing specific restoration activities (modified from Roni et al. 2002). Shaded boxes indicate where restoration actions should take place. Addition of salmon carcasses or nutrients may be appropriate at various stages following reconnection of isolated habitats.

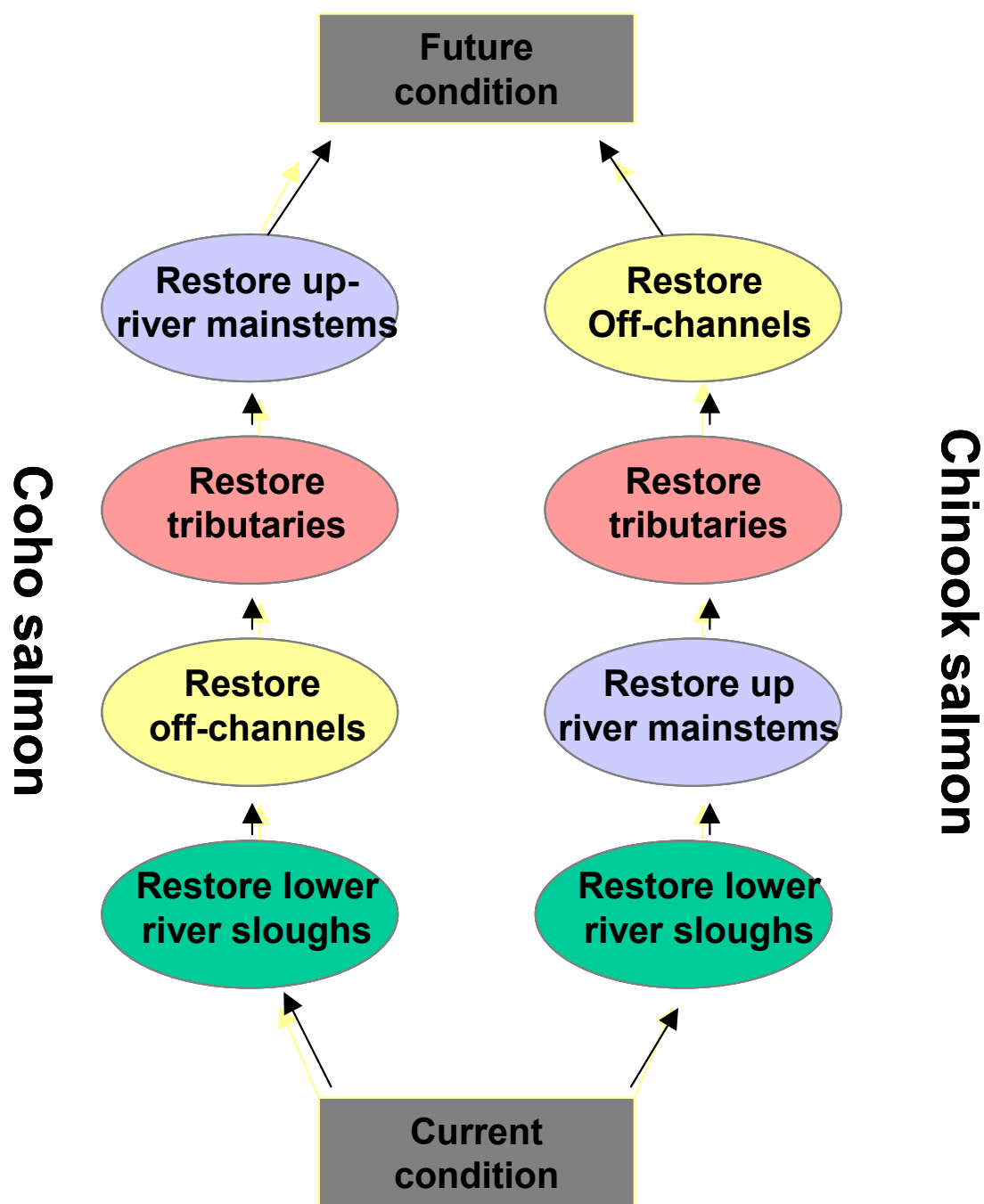


Figure 17. Sequence and prioritization of habitat restoration based on species of interest. Modified from Beechie and Bolton (1999).

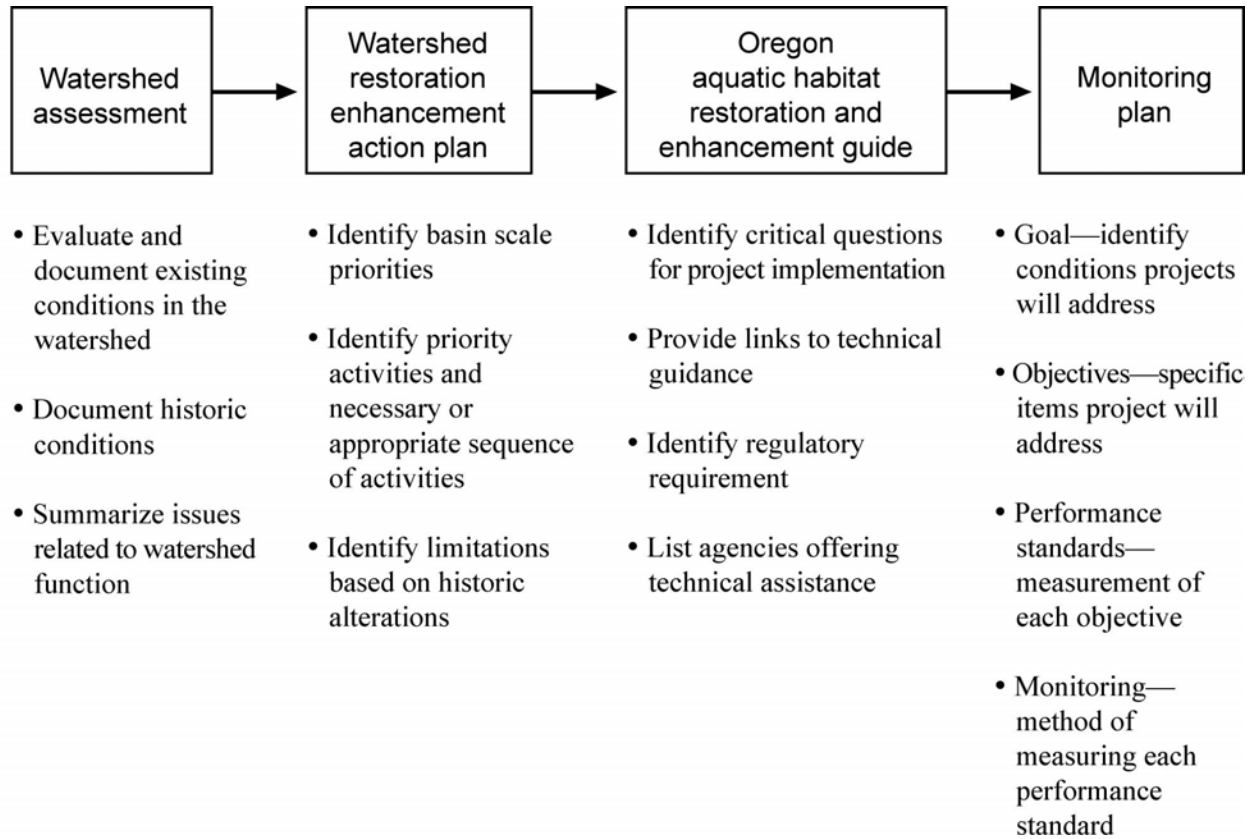


Figure 18. Process for restoration planning, prioritization, and implementation used by the Oregon Watershed Enhancement Board, Oregon Department of Fish and Wildlife, and state of Oregon. Figure reproduced from Oregon Aquatic Habitat Restoration and Enhancement Guide (OWEB 1999b).

MANAGING UNCERTAINTY IN HABITAT RECOVERY PLANNING

Salmon recovery planning requires a complex series of decisions about habitat actions despite large amounts of uncertainty in the available information. This uncertainty can result in risks to habitats and populations from inappropriate management advice (Fogarty et al. 1996). Past failures of management plans to prevent population declines and collapse are in part due to the failure to recognize uncertainty in available information and to a lack of procedures for including uncertainty in the decision-making process (Wade 2001). Inevitably, decisions will be based on a tapestry of models, estimates, expert opinion, myth, predictions, and data. By identifying, quantifying, and acknowledging the uncertainty in information used for recovery planning, we can increase the likelihood that recovery plans will be successful. The benefits of explicitly accounting for uncertainty include capturing all the available information regarding uncertain factors, providing the full range of possible outcomes and the probability of observing each, and identifying the key drivers of overall uncertainty in model projections (Mishra 2001). This section provides guidance via quantitative and qualitative examples for managing uncertainties inherent in habitat recovery planning.

A quick example illustrates how identifying and quantifying uncertainty can help a manager make explicit trade-offs between potential positive outcomes and acceptable risks. In choosing between two possible culverts for restoring fish passage, one might be presented with information that removal of culvert A is predicted to increase fish capacity by 120 fish while removal of culvert B is predicted to increase fish capacity by 100 fish. With no estimates of uncertainty, the manager would surely choose culvert A because it has the highest expected increase in fish capacity. Such a decision would be based on incomplete information. One also

needs uncertainty estimates, indications of how likely it is that the true outcome will be close to the point estimates of 120 and 100 fish. More complete information might indicate that replacement of culvert A would open habitat that was less certain to be occupied (120 ± 70), while replacement of culvert B would open wetland habitat that would be quickly colonized with a high degree of certainty (100 ± 10). With the more complete information, decision makers could then explicitly choose between a higher but less likely increase in fish capacity and a lower but more certain increase in fish capacity. In this example, both actions were unlikely to cause harm (a negative change in fish capacity). In other situations, actions with a high potential payoff may also contain some risk of being detrimental to fish, for example, when deciding whether to use chemical herbicides to remove non-native vegetation from riparian areas. Without an estimate of the magnitude of uncertainty in the information on which decisions must be made, decision makers cannot make informed decisions.

The importance of clearly communicating uncertainty has been repeatedly emphasized in the fisheries literature (Francis and Shotton 1997):

- “Understanding the risk or uncertainty associated with choices could help fisheries managers select management strategies, decide which types of risks and uncertainty inhibit the effectiveness of management techniques, and finally, recognize which types of uncertainty must inevitably remain...” (Peterson and Smith 1982);
- “Point estimates should be accompanied by variance estimates” (USCTC 1997);
- “The managers’ task may be made easier if uncertainty in a fishery assessment were expressed...” (Francis 1992);

- “Scientific advice to fishery managers needs to be expressed in probabilistic terms to convey the uncertainty about the consequences of alternative harvesting policies” (McAllister et al. 1994); and
- “Clearly, when management decisions are to be based on quantitative estimates from fishery assessment models, it is desirable that the uncertainty be quantified, and used to calculate the probability of achieving the desired target and/or risk of incurring undesirable events” (Caddy and Mahon 1995).

Reporting uncertainty in data and predictions has become common in harvest management (Rosenberg and Restrepo 1994). Uncertainty is not as often incorporated into decisions about salmon habitat recovery planning despite broad consensus that it is important and necessary to consider uncertainty in the conservation and management of species (Mangel et al. 1996, Flaaten et al. 1998, Akcakaya et al. 2000, Ralls and Taylor 2000, Wade 2001).

Additional information and improvements in decision making can be gained by partitioning the uncertainty into components such as measurement error versus model uncertainty. In the culvert example, partitioning of the uncertainty might reveal that the most significant source of uncertainty about culvert A is insufficient data on juvenile use of a few habitat types. By identifying knowledge gaps that limit decision making we can prioritize information gathering and model development. We emphasize that all information used in habitat analyses should include at a minimum both the observed or predicted value and an estimate of the associated uncertainty.

In this section, we first describe five types of uncertainty embedded in predictions of habitat capacity. We follow this with two examples of uncertainty in habitat management issues related to recovery planning. In each example, we describe how management decisions might be

improved by acknowledging, quantifying, and reducing uncertainty in the decision making process. The first example describes qualitative strategies for reducing uncertainties regarding chemical contaminants and for making structured decisions in the face of limited empirical data. The second example describes the use of decision tables for making decisions that incorporate uncertainty. The final subsection describes strategies for making decisions when empirical data are lacking. In this section, we distinguish between variability, which is characterized by differences in a variable's value over time, space, or populations, and uncertainty, which is lack of knowledge about a true and constant value of a quantity (Morgan et al. 1990, Cullen and Frey 1999). Our discussion of methods for reducing uncertainty is purposefully simplified throughout, but references are provided for each example so that interested readers can locate more detailed information. We hope that by omitting site-specific and mathematical details, a general framework for incorporating uncertainty into decisions can be expressed.

Types of Uncertainty

Precise and accurate predictions are a fundamental goal in the aquatic sciences. Improved management of aquatic resources will result from a predictive science that can forecast the consequences, costs, and benefits of management actions (Pace 2001). A prediction might be a predicted value (e.g. habitat capacity estimate, extinction risk, or survival rate) or a predicted relationship between a habitat action and a biological response (e.g. effects of high flows on egg survival, effects of a particular restoration technique on fish survival, or projected population trajectories under different climate scenarios). Population viability and habitat goals (Phase I Recovery Planning) as well as prioritized project lists and watershed plans (Phase II Recovery Planning) must be developed from these types of predicted values and relationships. Informed plans and decisions will be based on both the predictions and the uncertainty surrounding them.

The five types of uncertainty found in predictions of habitat capacity are predictive uncertainty, parameter uncertainty, model uncertainty, measurement uncertainty, and natural stochastic variation (Table 13).

Estimates of uncertainty are critical to informed decision making. Evaluating the relative magnitudes of the five types of uncertainty embedded in a particular prediction is valuable because it tells us where to be skeptical. More formally, we may pursue value of information (VOI) analysis to establish which additional information is most likely to improve our decision making position (Raiffa and Schlaifer 1961, Raiffa 1997). VOI techniques seek to identify situations in which the cost of reducing uncertainty is outweighed by the benefit of the reduction. In some cases, it may turn out that the predictive uncertainty is prohibitively large and, therefore, that the available empirical data provides little guidance for decision making. In such cases, other decision-making processes that do not require quantitative predictions can be used (see “Using Decision Rules When Empirical Data Are Inadequate” subsection, page XXX).

To a great degree, the five types of uncertainty are nested: prediction uncertainty includes parameter and model uncertainty, which each includes measurement error and natural variability. Here we start with the broadest form of uncertainty, prediction uncertainty, and work down to the underlying natural variation. We provide examples of how each type of uncertainty arises, how it might be quantified, and how it might be reduced (Table 13). We conclude each subsection with a summary of how decision making can be improved by quantifying and acknowledging each class of uncertainty. A series of questions to ask of any prediction is provided in Table 14.

Prediction Uncertainty

Predictions include uncertainty from natural stochastic variation of the system being modeled, measurement uncertainty of the data used to build the model, uncertainty surrounding the form of the model, and parameter uncertainty (components addressed in the following subsections). In addition, predictions can also include uncertainty that results from applying a model to a new situation. A capacity estimate for Watershed X might predict future capacity based on current and past data for the same watershed. Or current predictions for Watershed X might be based on data collected in other watersheds. Both cases involve extrapolating from conditions under which data were collected to new conditions of interest. Uncertainty associated with these or similar extrapolations from, say, the laboratory to the field, is difficult or impossible to quantify but must be considered and described.

Prediction uncertainty can be evaluated by ground-truthing (i.e., field measurement of specific attributes), by prediction confidence intervals, and by cross-validation simulation studies. Ground-truthing will help quantify the accuracy and precision of past predictions about current conditions, but can only suggest how well the model may perform under future conditions. Prediction confidence intervals can be computed in situations for which the manager does not need to extrapolate beyond the original data (Zar 1984). Where there is more than one predictor variable, caution should be used in defining the joint sample space beyond which one is extrapolating. In cross-validation simulations, the model is constructed and parameterized using a subset of the data (Stone 1974). The model is then assessed by how well it predicts that subset of data excluded from model construction. Cross-validation simulations do not include uncertainty associated with extrapolating from measured to unmeasured conditions. To assess how well a model may predict unmeasured conditions requires careful consideration of those

model components that may be sensitive to expected differences between measured and unmeasured conditions (i.e., current vs. future conditions). Models can be compared in their relative sensitivity to changing conditions. Models that rely on predictors that are only correlated with the causal factors are particularly likely to have high levels of prediction uncertainty because in new situations the correlations on which the model is based may no longer be coincident with the causal mechanism.

Parameter Uncertainty

Model parameters are necessarily estimated with uncertainty. A statement of the uncertainty of these parameter estimates is critical for making informed management decisions. Parameters that have biological meaning provide a context for interpreting the associated uncertainty. For example, if instream wood restoration were estimated to increase fry to smolt survival within the treatment reach by 20%, we might embark on a widespread wood placement plan. However, if the estimate were more completely expressed as 20% +/- 30%, we might diversify the types of restoration actions used, or choose a different restoration action with a smaller but more certain fish response and little or no risk of an adverse affect. For statistical models, parameter estimates are developed from the data and the uncertainty associated with these estimates is relatively easy to compute. For mechanistic models, parameters may be estimated from data, from similar models of other phenomena, or by expert opinion. Where parameters are not estimated from data, the uncertainty surrounding them can be difficult or impossible to quantify.

Sensitivity analyses can be used to estimate the effect of parameter uncertainty. Nominal range or local sensitivity analysis computes the effect on model outputs of systematically varying each of the parameters in the model across its range of plausible values while holding the other

inputs at their nominal values. Where small changes in parameter values lead to large changes in model predictions, the uncertainty of those parameters should be carefully evaluated. Models that are extremely sensitive to changes in parameter estimates and which have highly uncertain estimates of those parameters will yield predictions with large uncertainty. Even where models produce highly uncertain predictions, they may be useful in quantifying the uncertainty in predictions and in determining the type and quality of information that would be required to produce predictions with acceptable levels of certainty. The sensitivity analysis tells the manager that predictions are very sensitive to particular conditions and that they will either have to increase precision of parameter estimates or ensure that management plans are robust to expected uncertainty. Increased precision of parameter estimates can be achieved by collecting more data, collecting data over a wider range of values, or collecting better data (data with less measurement uncertainty).

Model Uncertainty

Nearly all estimates and predictions used in management are based on an underlying model, either explicitly or implicitly. Uncertainty exists about both the model form (for example a linear relationship vs. a Ricker curve) and about which predictor variables to include. Model uncertainty results from an incomplete understanding and a simplified representation of ecological systems and functions (Fogarty et al. 1996). In our capacity example, we might have a model that predicts habitat capacity as a linear function of several habitat parameters: wood density, pool density, gradient, adjacent land use, and water temperature. The default assumption may be to use a simple linear regression model. However, there is uncertainty as to whether the effects of these five habitat descriptors are additive or have a linear relationship to habitat capacity. The linearity assumption may be valid for the range of, say, water temperature

for which we have data, but invalid outside that range. We are also unsure if these five habitat descriptors are the best set of predictors or if an alternate set of predictors might perform just as well. Many statistical tools are available for choosing between models (adjusted R-squared, Akaike's information criteria or AIC, Bayesian information criteria or BIC, F-tests, likelihood ratio tests, cross-validation metrics) (Burnham and Anderson 1998). In general these techniques balance the degree to which the model fits or predicts the data with the complexity of the model, usually expressed as the number of parameters.

Models that fail to describe the ecological process accurately or to include an important predictor can have enormous management implications. Model predictions can be of the wrong magnitude or even the wrong direction. Managers and ecologists have often erred significantly by failing to consider model uncertainty. For example, the prevailing model of habitat effects on fish survival once assumed that fish survival decreases with increasing amounts of instream wood, and as a result, large amounts of wood were removed from streams and rivers (Maser et al. 1988). Thus habitat degradation in the Pacific Northwest can in part be attributed to a failure to assess the possibility that this model was incorrect (Beechie et al. 1996).

Model uncertainty is very difficult to quantify because there are an infinite number of possible models; none is exactly correct. Simulation studies generate data using a particular model, then ask questions about the behavior of those data (Morgan et al. 1990). They can quantify the degree to which the structure of the model influences the model's predictions. Averaging predictions from a suite of models can reduce the impact of model uncertainty on management predictions (Burnham and Anderson 1998, Cullen and Frey 1999). Beyond these tools, reducing model uncertainty is extremely difficult. Schnute and Richards (2001) suggest

that model uncertainty be managed by keeping an open mind, identifying all assumptions, and testing those assumptions continuously.

Measurement Uncertainty

Measurement uncertainty or observation error is simply the difference between a true value and our recorded observation of it. It results from measurement, sampling, and data processing errors (Francis and Shotton 1997). All observations carry some degree of measurement uncertainty. This uncertainty may be large and problematic or small and of negligible consequence. Some phenomena such as the survival of fish in different habitats are inherently difficult to measure. Consequently, the variables associated with these phenomena have a high degree of measurement uncertainty. Other phenomena such as stream discharge can be measured quite accurately. Uncertainty resulting from sampling error occurs when the measured units are not representative of the population for which inference is being made. The incorporation of measurement and sampling errors can obscure or create relationships between variables (Ludwig and Walters 1981, Walters and Ludwig 1981). Measurement error as defined here can also occur during data processing and storage.

Measurement uncertainty is directly related to both the accuracy and precision of the measurement technique. Accuracy in a measurement technique, the inverse of uncertainty, describes the average distance between the measured value and the truth. The precision of a measurement describes the variability around that average. Therefore, it is quite possible for a measurement tool to be highly precise (very low variance across repeated measurements) and yet inaccurate (the average of repeated measurements is far from the true value). In other words it is possible for a measurement to be characterized by very little variability but by a large degree of uncertainty. While there have been many attempts to estimate measurement uncertainty in, for

example, habitat surveys (Pleus 1995, Roper and Scarnecchia 1995, Poole et al. 1997) or redd surveys (Jones III et al. 1998, Dunham et al. 2001), the known uncertainty in these types of data is rarely included in the uncertainty of predictions from models that are based on these types of data.

Measurement uncertainty can result in systematic error or bias. Bias is a directional error that results from measurement using a systematically inaccurate tool. Biased or potentially biased measurements might include subjective assessments or incomplete records. A less visible form of bias occurs when a measurement technique tends to overestimate in certain conditions and underestimate in other conditions. A simple example is helicopter redd surveys. Redds are easier to identify where there are fewer trees; therefore the accuracy or uncertainty of the measurement depends on whether there are riparian buffers. If the bias is not corrected, the data can erroneously predict increases in redd density with removal of riparian trees.

Measurement uncertainty can be reduced but not eliminated. Replication is the best way to reduce the uncertainty, though it will not remove bias resulting from the use of inaccurate measurement tools. The best way to manage bias is to estimate it using at least some unbiased measurements and then correct for it. In some cases, measurement uncertainty can be very difficult to assess. Expert opinion or subjective assessments, for example, are often used because no actual data exists. Although it may be possible to determine how well experts agree with one another (precision), it is impossible to assess or quantify the relevant issue (accuracy) when there are no accurately measured data available for comparison. Measurement uncertainty may also be quantified using repeated measurements or by computer-intensive techniques such as resampling or bootstrap methods (Efron and Tibshirani 1991, Efron and Tibshirani 1993).

By quantifying measurement uncertainty, the value of collecting more data with the same measurement or sampling technique versus a more expensive technique can be weighed. Where bias is impossible to measure or quantify, sensitivity analyses (as just described in the Parameter Uncertainty subsection) can provide an assessment of the degree to which small amounts of measurement uncertainty or bias in the input data might effect predictions (Morgan et al. 1990).

Natural Stochastic Variation

Natural stochastic variation is the inherent random variability in ecological systems, such as temperature or population fluctuations. It also incorporates the underlying stochastic nature of population dynamics (Rosenberg and Restrepo 1994). It contributes to our inability to make precise predictions. Increased amounts of natural stochastic variation, often called process uncertainty, require increased numbers of observations (either more sites or more replications or both) to make estimates of a given precision (Shea and Mangel 2001). Very high levels of natural variation can mean that estimates of the required precision are simply impossible to obtain (Korman and Higgins 1997).

Identifying and quantifying natural stochastic variation helps us to distinguish between situations in which small amounts of additional data should dramatically increase our ability to make good decisions, and situations in which additional data are unlikely to provide significant increases in the accuracy of predictions. This is the heart of VOI analysis discussed earlier. While we may be able to improve on our knowledge of the true value of an uncertain parameter, we can only improve on our understanding of the behavior of a variable parameter. In some cases, stratifying the data or redefining the question can reduce the effects of natural stochastic variation. For example, we might make separate estimates of in-river survival for wet versus dry years. Managers would then be able to make more informed decisions about the value of habitat

restoration plans that potentially have different effects in wet versus dry years. Because stochastic variation is a natural phenomenon, it cannot be reduced to increase the precision of our predictions. Where it can no longer be reduced by stratification, quantifying and acknowledging stochastic variation is the best way to manage it.

In summary, an informed management decision requires information about the uncertainty of the predictions on which that decision will be based (Pace 2001, Regan et al. 2002). Evaluating the uncertainty in each prediction requires the dissection of that uncertainty into its components—quantifying and evaluating each and then recombining them into a single measure of overall uncertainty. Each class of uncertainty as well as methods for quantifying and reducing it are summarized in Table 13. By asking the questions in Table 14, we reduce the chances of making poor or uninformed decisions because of poor predictions. Regan et al. (2002) have identified further types of uncertainty collectively described as linguistic uncertainty, including concepts such as numerical vagueness, context dependence, ambiguity, and underspecificity. While we do not consider these forms of uncertainty explicitly in this section, they are certainly common in habitat science and should be sought out, evaluated and treated (Regan et al. 2002). Uncertainty analysis is key to identifying critical knowledge gaps, identifying options for improving predictions, and making the most informed habitat decisions.

Example 1: Creating a Prioritized List of Restoration Projects

Once we have a series of predictions with their associated uncertainties, we must combine them into an action plan (see “Prioritizing Potential Restoration Actions Within Watersheds” section, page XXX). In this example, we demonstrate one method of setting up a decision table for using predictions and their confidence intervals to develop a project list for a habitat recovery plan. Developing a project list is difficult because of uncertainty about how fish

may respond to changes in the environment. For example, we may have a list of potential actions, each of which is expected to increase pool habitat. There are uncertainties in estimating the increase in pool area and uncertainty about the density of fish that can be supported by a given amount of pool habitat. By explicitly including the uncertainty in a decision table, we can identify the actions with the highest expected final fish density and determine the potential value of reducing the uncertainty. Analogous examples have been worked out in the harvest literature (Hilborn and Walters 1992).

The first task in setting up a decision table is to describe the “alternative states of nature” and ascribe probabilities to these states. In this example, the alternative states of nature are the alternative hypotheses about how many juveniles are supported by a given area of pool habitat. Table 15 presents sample hypotheses and associated probabilities. The probabilities associated with each hypothesis may be generated in a number of ways. One method that can combine multiple types of information is meta-analysis, which pulls together information from multiple sources (Liermann and Hilborn 1997, Myers et al. 2001). Other Bayesian analysis techniques can also be used to combine disparate sources of information. A trademark of Bayesian analysis is the assignment of probabilities to alternative states of nature (Wade 2000). Strengths and weaknesses of the Bayesian approach are described in a special section of Ecological Monographs edited by Dixon and Ellison (1996). If only limited or ambiguous data are available, expert opinion can be solicited to assign probabilities to the various hypotheses. Numerous texts describe methodologies for soliciting expert opinion (Morgan et al. 1990, Cooke 1991). Readers are encouraged to explore these for full descriptions of the complexity of selecting a group of experts, combining their disparate judgments, and other challenges of this approach. As noted elsewhere in the section, it is impossible to know if expert opinion is correct

precisely because we use it in situations for which we have no data. If expert opinion is used to assign probabilities to a set of hypotheses, then the prioritized list that emerges from the decision-analysis process will be a formalization of those opinions.

The next step in setting up a decision table is to associate an outcome with each potential action, assuming each of the alternative hypotheses about the state of nature is true. For example, if the hypothesis that pools can support five juvenile fish per m² is true, then the number of fish expected from the removal of culvert A might be 2,744 fish. In this example the outcome is number of fish, but other appropriate outcome units, such as fish per dollar, may be of interest. This outcome is calculated based on an assessment of the number of pools that would be made available after removal of the culvert. More realistic and detailed decision tables might include additional information such as the number of riffles, types of pools, depths of pools, or quality of expected pool habitat. Table 15 shows potential outcomes in total fish for a number of management actions as a function of fish density in pools.

Finally, we calculate the final expected outcome of each of the potential actions, given the probabilities of the states of nature (Table 15). The expected outcome of each action is calculated by summing the expected outcome for each state of nature multiplied by the probability that the state of nature is true. For example, the expected outcome for removal of culvert A is $2744 * 0.1 + 4892 * 0.3 + 5248 * 0.5 + 5786 * 0.1 = 4945$. Table 15 shows the expected outcome for each of the four potential actions. The largest expected increase in total number of fish is associated with removal of culvert A.

This is an extremely simple example. Hypotheses about the states of nature will often involve more than a single dimension (e.g., more than pool density). Many types of information can be included in the analysis, but there will often be only one or two critical uncertainties that

drive a decision. Decision tables provide a structured method for including and communicating uncertainties and can easily be constructed for many of the examples in this document. For example, the methods described in the “Prioritizing Potential Restoration Actions Within Watersheds” section, page XXX, could be modified to include uncertainty about fish response, restoration costs, or habitat quality by using the decision table methodology described here. Another tool for making decisions is a logic tree, which models the impact of uncertainties in states of nature and in the occurrence of future conditions on possible outcomes (Kessler and McGuire 1999). Logic trees are particularly useful when only subjective probabilities about the states of nature exist.

Example 2: Water Quality and Habitat Recovery Planning

Uncertainty in habitat planning can result from the omission of a key habitat variable, such as water quality. The quantity and quality of salmon habitat are both important determinants of salmon population viability. Stream temperatures, sedimentation, and water pollution are all examples of habitat quality. Importantly, however, empirical data for the various forms of water pollution are rarely incorporated into habitat models. Consequently, the complex impacts of urbanization, agricultural land uses, and industrial activities on the chemical condition of salmon habitat may lead to large levels of uncertainty in habitat recovery planning. In this example, we suggest ways to improve habitat decision making by incorporating water quality data. We provide non-quantitative solutions to reducing uncertainties that result from the omission of key habitat variables.

Environmental monitoring studies have consistently detected a wide array of metals, pesticides, and other toxic substances in the surface water and sediment of salmon habitats, and also in the tissues of salmon themselves. These contaminants may affect salmon abundance and

survival via immediate lethal effects on individual fish. However, such effects are rare compared to the vast array of potential sublethal effects that may reduce individual fitness and population performance and of potential indirect effects such as reductions in the abundance of key prey taxa. Despite documented exposure conditions (Wentz et al. 1998, Ebbert and Embrey 2002), the impact of environmental contaminants on salmon health or on the biological integrity of aquatic systems is poorly understood and habitat-based models for salmon recovery rarely capture the biological significance of water and sediment quality. Predictions of salmon population viability are likely to have high levels of model and prediction uncertainty if water and sediment quality are not included in model development.

There are several reasons why the specific determinants of chemical habitat quality are often excluded from habitat models. First, chemical habitat quality can be difficult and expensive to measure. Second, there is a general absence of toxicological data for most of the chemicals that have been detected in salmon habitat. Third, many conventional endpoints or biomarkers of chemical exposure have no clear or consistent relationship to the survival or reproductive success of the exposed animal. Consequently, there is often a disconnect between the biological scale at which toxicological studies are conducted and the data requirements for current habitat recovery models (Hansen and Johnson 1999a, 1999b).

Recovery plans that capture broad spatial and temporal patterns of chemical habitat degradation, despite incomplete empirical data, will minimize uncertainties around predicted outcomes of restoration actions and therefore reduce risks to salmon populations. Contaminants occur in complex mixtures whose composition varies in time and space. Salmon habitat conditions may reflect current land use activities or activities that were restricted or banned many years ago (e.g., for persistent chemicals such as DDT). Moreover, water quality at a specific

point within a watershed may be determined by land use activities that are far removed from the focus of restoration efforts. Acknowledging the large spatial and temporal scales at which contaminants can affect fish helps identify some of the uncertainty associated with predicting the effects of restoration actions. We can surmise, for example, that the uncertainty of predicted increases in habitat capacity for a given restoration action is likely higher in areas with high levels of past or present on-site or upstream chemical contamination. Likewise, we might expect inaccuracy and prediction uncertainty in survival estimates that are extrapolated from a stock within a pristine watershed to a stock that migrates through a highly contaminated estuary.

In many cases, we do have data on chemical contamination but we do not know how to locate it or to incorporate it into habitat recovery planning. A limited number of studies have specifically addressed the impacts of environmental contaminants on biological processes in Pacific salmon that are clearly linked to survival, migratory success, or reproductive success (Kruzynski and Birtwell 1994, Arkoosh et al. 1998, Hansen et al. 1999, Heintz et al. 2000, Scholz et al. 2000, Rice et al. 2001, Meador et al. 2002). The challenge in estimating the effects of toxic chemicals on salmon health is to identify which contaminants are known or suspected to occur in particular habitats and pathways of toxicity for these chemicals that have significance for the survival, migratory success, or reproductive success of wild salmon.

Planners or researchers should utilize the primary toxicological literature in the development of recovery plans. Answers to the following questions can often be found in the toxicological literature and will enable more accurate and precise predictions about the effects of specific chemical contaminants on predicted salmon population performance. What is the evidence that a contaminant or class of contaminants is present in salmon habitat? What are the expected environmental concentrations? How long will exposures last? What life history stages

of salmon are likely to be affected? What are the primary possibilities for sublethal toxicity in fish? From this information it may be possible to estimate the chances that the contaminant is currently or may in the future be a significant limiting factor in salmon population viability within the geographical area of concern.

Incorporating toxicological data can improve decisions about the prioritization of water quality improvements versus physical habitat restoration. For example, in watersheds where insecticides occur (primarily in agricultural and urban areas), it should be possible to estimate the potential loss of invertebrate prey, the subsequent reduction in the growth of juvenile fish, and the likelihood that salmon from contaminated habitats will have a lower rate of marine survival. If environmental monitoring data are unavailable, recovery planners might extrapolate potential chemical concentrations from other (monitored) basins with similar agricultural or urban land use. Even simple comparisons between reported environmental concentrations and toxicity thresholds for aquatic invertebrates can reduce the scientific uncertainty surrounding the potential effects of contaminants on salmon population viability. This, in turn, would improve restoration prioritization and watershed management plans.

For water quality and other habitat characteristics about which less is known, it is clearly better to acknowledge the uncertainties and incorporate the available information, no matter how limited. In the example of water quality, we can estimate and incorporate the direction of the effect even when we are not yet able to quantify the magnitude of that effect. We can also seek empirical data from non-traditional sources. Moreover, identifying key uncertainties will help establish priorities for ongoing and future research.

Using Decision Rules When Empirical Data Are Inadequate

A careful and honest examination of uncertainty in data, predictions, and models will inevitably lead to the identification of situations in which adequate empirical data for making a decision are simply not available. Uncertainty should not lead to inaction. Methods are being developed to allow quantitative analysis of the sensitivity of decisions to uncertainties in the data. For example, sensitivity analyses were used to demonstrate that the best management decision for Hector's dolphin (*Cephalorhynchus hectori*) was robust to model uncertainties, and thereby removed uncertainty in the scientific data as an excuse for inaction (Slooten et al. 2000). In the face of large amounts of uncertainty in empirical relationships, simulation models and decision analysis were used to evaluate management actions for listed salmonids in the Snake River basin (Peters and Marmorek 2001, Peters et al. 2001). Where empirical data are inadequate, we strongly discourage basing decisions on biased or imprecise predictions, prioritization systems for which guesswork must be substituted for data, or information that becomes inaccurate or imprecise at the scale for which the decision must apply. Instead, we suggest that managers provide an explicit rationale for the decision that requires minimal data.

The most important characteristics of a decision rule are that it can be documented and that it is robust. Documentation is important because future managers will need to understand the basis for the decision. This requirement prevents arbitrary decisions in the face of inadequate data. Decision rules that are robust to the uncertainties in the information prevent risky management decisions (Schnute and Richards 2001). Decision rules that have been presented in the literature include the following two examples.

The Precautionary Principle can be stated as, "When an activity raises threats of harm to public health or the environment, precautionary measures should be taken even if some cause

and effect relationships are not fully established scientifically” (Raffensperger and Tickner 1999). Because this principle shifts the burden of proof to those who create risks and does not define which risks are most important (Hilborn et al. 2001), it has generated much controversy and confusion about its appropriate implementation. However, there are many examples of national and international policies that have been based on the Precautionary Principle.

European environmental law is based on the precautionary principle through the Treaty on European Union, 1992 and international treaties, including the Rio Declaration from the United Nations Conference on Environment and Development, bind the United States to implement the Precautionary Principle in environmental health protection (Raffensperger and Tickner 1999). While we are not advocating this particular decision making rule, we present it as an example of a relatively simple guiding principle for high-level decisions in the absence of definitive data.

Safe Minimum Standard (SMS) is another decision-making rule that has received considerable attention. The SMS approach is a collective choice process that prescribes protecting a given level of a renewable resource unless the social costs are excessive (Berrens 2001). This approach to making environmental decisions is usually invoked in settings involving considerable uncertainty and potentially irreversible losses. It prioritizes social costs over loss of renewable resources. We present this approach for comparison to emphasize the importance of carefully choosing the decision-making principle and documenting exactly what considerations should be involved. The choice of a guiding principle will dictate management decisions until improved information is available.

The choice of a decision-making rule need not be purely theoretical. The “An Assessment Approach for Habitat Recovery Planning” section, page XXX, discussed the importance of defining a habitat strategy that includes gathering additional data and taking

interim actions. This habitat strategy is an excellent example of how a guiding principle can be used for decision making until adequate data become available. The “Prioritizing Potential Restoration Actions Within Watersheds” section, page XXX, presented guidelines for selecting restoration actions before all of the habitat data are available. Again, this is a simple and effective method for dealing with incomplete information.

Another common approach to formalizing decision making without adequate empirical data or quantitative predictions is a scoring matrix. A scoring matrix can be used to prioritize potential actions, project proposals, potential action sites, or information gathering. The advantage of a scoring matrix is that ranks can be based on weighted priorities, for example project longevity, proximity to other projects, or land ownership. The decision path can be clearly explained and is easily repeatable. As better information becomes available, the matrix can be adjusted. A disadvantage of scoring matrices is that the weights assigned to each priority can dramatically alter the outcome and specifying a satisfactory weighting function in advance is often difficult. Examples of scoring matrices in current use include the Snake River Salmon Recovery Region Comprehensive Project Scoring Matrix (SRSRC 2002), the Lower Columbia Fish Recovery Board Interim Habitat Strategy Project Scoring Sheet (LCFRB 2001), and the Skagit System Cooperative methodology for rating individual landscape processes (Appendix C, page XXX). The scoring matrix provided by the Lower Columbia Fish Recovery Board dedicates a section to “Certainty of Success,” explicitly including some metrics of uncertainty.

In each of the above examples, it is important to consider whether the decision strategy is robust to the types of uncertainties that exist. A strategy that would be beneficial under a scenario that has a 50% chance of representing reality but that would be detrimental the rest of the time is not a robust choice. Strategies should be developed so that the outcome is acceptable

given the range of possibilities for which there is uncertainty. Again the Hector's dolphin management plan is an example of a strategy that is explicitly robust to the uncertainties in the data (Slooten et al. 2000).

Using decision-making strategies that require minimal data carries two obligations. First, we must evaluate whether improved information would produce a cost-effective improvement in decision making (value of information analysis). If so, then a strong attempt to reduce uncertainties by gathering more or better information is required. The analyses described in the first example of this section (evaluating a prediction) can be used to identify critical information uncertainties and reduce their impact. Second, we must set a time frame for reevaluating the decision. In the best possible scenario, decision strategies requiring minimal data serve as interim measures until additional information is available.

In conclusion, we emphasize that estimates of uncertainty—quantitative where possible, qualitative for other situations—should be included with all information being considered in a decision-making framework. A systematic treatment of uncertainty should include: 1) identification of uncertain events, states of nature, relationships, and parameters; 2) determination of the likelihood associated with each potential state or value; 3) use of data or models to evaluate consequences of each potential state or value; and 4) examination of the relationship between uncertain inputs and potential outputs to identify key uncertainties (Mishra 2001). Even where a formal decision theoretic approach is not possible, describing sources and magnitudes of uncertainty is important in providing managers with enough information to weigh potential risks and benefits of possible actions (Rosenberg and Restrepo 1994).

A careful examination of the sources and causes of uncertainty will ensure informed decisions and make improvements in both precision and accuracy likely. Quantifications of

uncertainty can be formally incorporated into decision making using decision tables. In other situations, simple strategies such as collecting data at multiple scales or incorporating data from other disciplines will provide for more informed decisions. However, a lack of empirical data need not prevent informed decisions from being made in a clear and formal manner. It is possible to implement strategies that require minimal data. Such strategies are preferable to using biased or imprecise predictions, guesswork disguised as data, or information that is inappropriate to the scale of the decision.

As we have said in this technical memorandum, our conceptual approach to habitat recovery planning is holistically focused on restoring or preserving watershed and ecosystem processes to provide good quality salmon habitat over the long term. This implies that restoration of ecosystems to support salmon will include a wide range of recovery actions affecting the life cycles of multiple species. We began with a conceptual framework for understanding relationships among land uses, watershed functions, habitat conditions, and biota as a basis for organizing the habitat-related questions that each recovery plan should attempt to answer. We separated recovery planning into two phases—Phase I planning that identifies recovery goals and Phase II planning that identifies causes of habitat loss or degradation and necessary ecosystem restoration actions. Then we showed how results from both assessments can be used to prioritize restoration actions and how estimates of uncertainty can be incorporated into the planning. Incorporating estimates of uncertainty into the decision-making process increases the likelihood of success in salmon habitat recovery planning. Remember that new information gained from future assessments and management experiments should be used to update the recovery plan in the future.

Three appendices follow in this guidance document. Appendix A, Issues of Scale in Habitat Recovery Planning, examines the concept of scale in recovery planning with particular emphasis on analyses to help set recovery goals. It demonstrates how the effects of habit change on the four components of population viability can be examined over multiple scales and includes examples from the literature to illustrate the types of information obtained at each scale. Appendix B, Estimating Chinook Salmon Spawner Capacity of the Stillaguamish River, is an example analysis for Phase I recovery planning. This case study estimated the river's historical and current capacity for adult chinook salmon based on habitat data at the unit scale (e.g., pool, riffle, and glide), then extrapolated to the watershed scale to estimate the maximum number of adult chinook that the river produced historically and the system's potential for production today. Appendix C, Restoration of Habitat-forming Processes: An Applied Restoration Strategy for the Skagit River, is an example analysis for Phase II recovery planning. It briefly describes the Skagit Watershed Council's habitat protection and restoration strategy, directed at restoring the disturbed habitat-forming processes instead of attempting to build specific habitat conditions, as well as applications of the methods and preliminary results.

Table 13. Tools and methods for quantifying and reducing uncertainty.

Class of uncertainty	Brief definition	Habitat example	Methods for quantifying	Possibilities for reducing
<i>Prediction uncertainty</i>	Difference between the modeled response and the true response.	Uncertainty of predicting habitat capacity of a given watershed after instream restoration.	Leave-one-out estimates of prediction error rates. Simulation studies comparing conditions where model was built to those in which it is being applied.	Collect data for conditions in which predictions are required. Do not extrapolate beyond conditions under which model was developed.
<i>Parameter uncertainty</i>	Difference between the true parameter (such as an average or a regression coefficient) and the parameter as estimated from the data.	Uncertainty of parameters describing change in capacity as a function of changes in watershed condition.	Statistical theory for model coefficients derived from data. Sensitivity analysis for model coefficients estimated from other sources.	Collect more data or more accurate data. Collect data over a wider variety of conditions.
<i>Model uncertainty</i>	Difference between natural system and the mathematical equation used to describe it. Includes model form and set of predictors.	Uncertainty in the relationship between habitat conditions and fish capacity. Uncertainty in which habitat descriptors are the best predictors of fish capacity.	Statistical descriptions of model fit: Akaike's information criteria (AIC), Bayesian information criteria (BIC), likelihood ratios, F-statistics.	Consider wide variety of models. Sensitivity analyses.
<i>Measurement uncertainty</i>	Difference between true value and the recorded value.	Uncertainty in measurements of data used to build the predictive model, i.e. fish or redd density under differing habitat conditions.	Test accuracy of measurement technique against standard method or known values.	Improve measurement techniques. Increase the number of replicates. Calibrate biased measurement techniques.
<i>Natural stochastic variation (process uncertainty)</i>	Inherent random variability.	Natural fluctuations in population size, habitat selection, or habitat conditions.	Variance of the observed data. Variance of the observed data for different sets of conditions.	Collect more replicates for conditions of interest. Stratify data collection.

Table 14. Questions to guide the evaluation of predictions.

Prediction uncertainty

How similar are the conditions under which the original information was gathered to those for which the prediction is being made? How sensitive is the model (data, mechanism, and parameter estimates) to site-specific details?

Parameter uncertainty

Is the prediction sensitive to small changes in parameter estimates? If so, how precise are the estimates of those parameters?

Model uncertainty

What are the assumptions on which the prediction is based? How sensitive is the prediction to these assumptions?

Measurement uncertainty

Could any of the information on which the prediction is based be biased? How precise and how accurate are the data?

Natural stochastic variation (process uncertainty)

Can measurements be stratified across conditions to reduce the effects of natural variability?

Table 15. Input information and results of decision analysis for prioritizing restoration actions. Example alternative hypotheses about the states of nature (i.e. density of fish per m² of pool habitat) and the relative probability that the hypothesis is true are in the first two rows. All probabilities must sum to one. Expected outcomes for potential habitat actions (total fish) as a function of each of these hypothesized fish density are displayed below the hypothesis probabilities. Overall expected outcomes (increase in total number of fish) of each potential action, given all potential states of nature, are in the last column.

Hypothesized fish density per pool		5	10	15	20	Overall expected outcome
Hypothesis probability		0.1	0.3	0.5	0.1	
Potential Action	Remove culvert A	2,744	4,892	5,248	5,786	4,945
	Remove culvert B	2,844	3,400	3,858	6,457	3,879
	Remove riprap	2,012	4,172	4,260	4,340	4,017
	Add wood	1,568	3,410	5,963	6,230	4,784

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GLOSSARY

Anadromous

Moving from the sea to fresh water for reproduction.

Anthropogenic

Caused or produced by human action.

Bankfull width

Channel width between the tops of banks on either side of a stream; tops of banks are the points at which water overflows its channel at bankfull discharge. (Compare *wetted width*.)

B-IBI

Benthic Index of Biological Integrity: An overall assessment of invertebrate condition constructed from various biometrics and represented by a single number. (See *biological integrity* and *IBI*.)

Benthic

Of, related to, or living in the soil-water interface of a lake or stream.

Biodiversity

The range of different species of plants and animals in an environment or during a specific period of time.

Biological integrity

Defined in various ways, here it is the capability of supporting and maintaining a balanced, integrated, adaptive community of organisms having a species composition, diversity, and functional organization comparable to that of the natural habitat of the regions (Karr 1991).

Biota

The flora and fauna of a region.

Culvert

Buried pipe or covered structure that allows a watercourse to pass under a road or underground.

CWA

Clean Water Act: Passed by Congress in 1972, its purpose is to “restore and maintain the chemical, physical, and biological integrity of the nation’s waters.”

Distributary (also distributary channel)

A branch of a river or stream that flows away from the main channel and does not rejoin it.

Disturbance

Introduction of an unwanted condition into a system or interference with a habitat’s normal or existing conditions.

Ecoregion

An area determined by similar land surface form, potential natural vegetation, land use, and soil; it may contain few or many geological districts.

Ecosystem

In general use, it is the dynamic and holistic system of all the living and dead organisms in an area and the physical and climatic features that are interrelated in the transfer of energy and material. In this document, it is the aquatic environment and biota, physical and biological processes active in that environment, and the landscape processes and land uses that form and sustain the aquatic environment and biota. (Compare *habitat*.)

EDT

Ecosystem diagnosis and treatment model: An analytical tool for assessing relationships among stream habitat attributes and salmon population performance.

EIA

Effective impervious area. (See *impervious surface*.)

Endangered species

A species in danger of extinction throughout all or a significant portion of its range. (See *ESA*.)

Enhancement

Improving watershed processes and habitat conditions from an existing state. It does not necessarily seek to restore processes or conditions to some predisturbed state. Some practitioners call this partial restoration. (Compare *rehabilitation* and *restoration*.)

ESA

U.S. Endangered Species Act: Passed by Congress in 1973, its purpose is to “provide a means whereby the ecosystems upon which endangered species and threatened species depend may be conserved.”

Estuary

Semi-enclosed coastal body of water at the mouth of a river where the saltwater ocean tide meets the freshwater current.

ESU

Evolutionarily significant unit: A population or group of populations that are 1) substantially reproductively isolated from other populations, and 2) contribute substantially to the ecological or genetic diversity of the biological species (Myers et al. 1998). It is sometimes represented as the spatial area encompassing the population(s).

Floodplain

Flat area adjoining a river channel constructed by the river in its present climate and overflow at times of high discharge.

Fry

The brief transitional stage of recently hatched fish that spans from absorption of the yolk sac through several weeks of independent feeding.

Gabion

Wire basket filled with stones, used for enhancing aquatic habitats or stabilizing streambanks.

GIS

Geographical Information System: A computer system for assembling, storing, manipulating, and displaying geographically referenced information.

Glide

Relatively slow and shallow stream section with moderate water velocities and little or no surface turbulence.

Ground-truthing

Field measurement of specific attributes that have been predicted from models or remotely sensed data for the purpose of assessing accuracy and precision of predictions.

Gullying

Erosion of soil by formation or extension of gullies from surface runoff.

Habitat

In this document, the term refers to the aquatic environment that fish experience and not those landscape processes or attributes outside streams that alter habitat conditions. (Compare *ecosystem*.)

Habitat unit

Relatively homogenous area of the stream channel that differs from adjoining areas in depth, velocity, and substrate characteristics.

Hydromodification

Alteration of stream banks or channel morphology by bank hardening (e.g., riprap), dredging, diking, or other mechanical means.

IBI

Index of biological integrity: A synthesis of various biometrics that numerically assesses associations between human activities and biological attributes. (See *B-IBI* and *biological integrity*.)

Impervious surface

Defined in watershed management as man-made surfaces such as asphalt or concrete paving that prohibit the movement of water from the land surface into the underlying soil.

Impoundment

A body of water gathered or enclosed (such as in a reservoir) for irrigation, flood control, or similar purposes.

Landsat

Landsat satellites supply global land surface images.

LWD

Large woody debris: A large piece of woody material, such as a log or stump, that intrudes into a stream channel. Typically, LWD is defined as wood greater than 10 cm in diameter and 1 m in length, but other minimum size criteria are also used.

Main stem

Principle stream or channel of a stream network.

Mass wasting

Downslope movement of earth materials under gravity, including such processes as rockfalls, landslides, and debris flows.

Natural stochastic variation

The inherent random variability in ecological systems, such as temperature or population fluctuations. Also called process uncertainty.

Outlier

In statistics, any data point exhibiting anomalous behavior.

Peak flow

Greatest stream discharge measured over a period of time, such as a season or year.

Parr

A young salmonid actively feeding in freshwater.

Population

A group of individuals of a species living in a certain area that maintain some degree of reproductive isolation.

Pool:riffle:glide ratio

Ratio of the respective surface areas or lengths of pools to riffles to glides in a given stream reach, often expressed as the relative percentage of each category.

Reach

A section of stream between two defined points.

Recovery

In the context of listed populations, attaining specified goals for viable populations and ESUs (population abundance, population growth rate, spatial structure, and diversity). For watershed processes and habitats, returning from a disturbed state to some prior condition, not necessarily pristine.

Redd

Nest in gravel, dug by a fish for egg deposition, and associated gravel mounds.

Refugia

Also commonly called biological hot spots, source watersheds, core areas, and key habitat, they are population centers or centers of biological diversity.

Rehabilitation

Improving ecosystem conditions, or sometimes more specifically, returning ecosystem conditions to a defined level of health. Some practitioners call this partial restoration. (Compare *restoration* and *enhancement*.)

Remotely sensed data

Data gathered from a remote station or platform, as in satellite or aerial photography.

Restoration

In the strictest sense, returning the ecosystem to some predisturbed condition. Some practitioners call this full restoration. (Compare *rehabilitation* and *enhancement*.)

Riffle

Shallow section of a river or stream with rapid current and surface turbulence.

Riparian

The part of the landscape that exerts a direct influence on stream channels or lake margins and the water or aquatic ecosystems.

Riprap

Layer of large, durable materials used to protect a stream bank from erosion; may also refer to the materials used, such as rocks or broken concrete.

Salmonid

Fish of the family *Salmonidae*, including salmon, trout, and chars.

Scale

In aquatic environment recovery planning, it can be viewed as a hierarchy of spatially nested systems. Each of these spatially nested systems can be thought of as an individual spatial and temporal scale. The result is a system in which development and persistence occur at specific temporal scales within each level of the hierarchy such that conditions within smaller scale systems are constrained by the larger scale systems within which they are contained (Frissell et al. 1986, Urban et al. 1987).

Sediment budget

An accounting of sediment sources and transfer processes in watershed. The complete budget quantifies sediment sources, transport, and storage within a watershed, usually tracking each process of sediment production or movement separately.

Sediment supply

Supply of sediment to a river system, where it is carried in suspension (suspended load) or on the bottom (bed load).

Seral

Of, relating to, or constituting a sere (which is a series of ecological communities formed in ecological succession.)

Side channel

Flood channel or abandoned stream channel connected to a stream or river at periods of high flow. Serves juvenile fish as rearing habitat and refuge from floods.

Smolt

Juvenile salmonid in its seaward migrant stage.

Species

A category of biological classification ranking immediately below genus or subgenus, comprising related organisms or populations potentially capable of interbreeding. (Note that in the context of the ESA, “distinct population segments” of a species may be listed separately.)

Stochastic

Of or relating to uncertainties or random variables.

SWAM

Salmonid Watershed Analysis Model: A large scale landscape analysis for identifying high priority areas for salmon habitat restoration.

Taxa

Plural of taxon, a taxonomic group or entity.

Thalweg

The line defining the lowest points along the length of a river bed or valley.

Tributary

A stream or river that flows into another stream or river.

TRT

Technical Recovery Team: The TRT is responsible for establishing biologically based ESA recovery goals for listed salmonids within a given recovery domain. Members serve as science advisors to the recovery planning phase.

Threatened species

A species not presently in danger of extinction but likely to become so in the foreseeable future. (See *ESA*.)

Uncertainty

Lack of knowledge about the true value of a quantity or lack of knowledge about which of several alternative models best describes the mechanism of interest. Types of uncertainty include prediction uncertainty, parameter uncertainty, model uncertainty, measurement uncertainty, and natural stochastic variation (also called process uncertainty).

Variability

Heterogeneity of values over time, space, or different members of a population.

VSP

Viable salmonid population: an independent population of any Pacific salmonid (genus *Oncorhynchus*) that has a negligible risk of extinction due to threats from demographic variation,

local environmental variation, and genetic diversity changes over a long time frame (McElhany et al. 2000).

Watershed (also drainage basin or catchment)

The land drainage area of a stream.

Wetted width

Width of the water surface within a channel. (Compare *bankfull width*.)

APPENDIX A. ISSUES OF SCALE IN HABITAT RECOVERY PLANNING

The aquatic environment is complex and dynamic, changing continually across space and time. Inhabitants of this environment have evolved in response to these ever-changing conditions. However, anthropogenic alterations to the landscape have disrupted the natural processes within these systems and species are often forced to contend with altered or unnatural habitat conditions. These alterations can be large or small, influencing expansive areas or more local conditions, and the effects can occur immediately or years later. Thus, a thorough knowledge of the processes structuring the aquatic environment and how these processes interact over multiple spatial and temporal scales is critical for understanding the effects of disturbance on aquatic systems and their inhabitants.

This appendix examines the concept of scale in recovery planning with particular emphasis on analyses to help set recovery goals. Similar concepts apply to analyses designed to identify ecosystem recovery actions (described in the “Analyses for Phase II Recovery Planning: Identifying Ecosystem Restoration Actions” section, page XXX), but are not discussed. This appendix first describes the inherent hierarchical nature of aquatic systems, specifying the need to study both the processes and inhabitants of these systems in a similar hierarchical manner. Next, examples of small- and large-scale studies highlight difficulties in transferring information across scales. Finally, these concepts are incorporated into examinations of how habitat alterations might influence the four categories of recovery goals: population abundance, population growth rate, spatial structure, and diversity (McElhany et al. 2000). Examples from the literature illustrate the types of information obtained at each scale and demonstrate the value

of combining studies across scales to provide information on both impaired processes and recovery potential.

Hierarchical Nature of Stream Systems

The aquatic environment can be viewed as a hierarchy of spatially nested systems (Frissell et al. 1986, Urban et al. 1987). Implicit in this hierarchical model is the concept of scale. For example, reaches (10^1 m) are contained within watersheds (10^3 m), and physical conditions within reaches are driven largely by frequent (10^1 - 10^0 years), low magnitude geomorphic events such as floods (Frissell et al. 1986). By contrast, watershed attributes are influenced by infrequent (10^6 - 10^5 years), high magnitude geologic events (e.g., glaciation, tectonic movements). The result is a system in which habitat development and persistence occur at specific temporal scales within each level of the hierarchy, and conditions within smaller scale systems are constrained by the larger scale systems within which they are contained (Frissell et al. 1986, Urban et al. 1987).

In addition, longitudinal (upstream to downstream) and lateral (stream to terrestrial) linkages help shape biological and physical structure at each level resulting in a predictable spatiotemporal gradient of physical and biological conditions from headwaters to mouth (Vannote et al. 1980, Frissell et al. 1986, Gregory et al. 1991). For example, stream width, depth, discharge, (Platts 1979, Leopold et al. 1964), temperature (Allan 1995), and biological diversity (Vannote et al. 1980, Barila et al. 1981) increase with stream size while gradient and substrate size decreases (Platts 1979, Leopold et al. 1964). There is often a decrease in terrestrial inputs and riparian shading coupled with an increase in organic transport from headwaters to estuary (Vannote et al. 1980). Moreover, these conditions are structured by the climate, geology, and anthropogenic activity of the specific watershed (Vannote et al. 1980, Frissell et al. 1986).

As a result, each segment within the system contains a predictable array of habitat conditions dependent on the watershed. These habitats, however, are not homogenous—there is simply an order to their heterogeneity (Frissell et al. 1986).

Because habitats are heterogenous, species distributions are not even across the landscape but instead occur in patches. The quality and quantity of each habitat is a product of both large- and small-scale processes, and quantifying this environmental variation helps to understand subsequent biotic responses (Cunjak 1996). For example, water temperature varies both spatially and temporally and influences salmonid distributions and life-history patterns. Large-scale temperature patterns influence species ranges and distributions (Meisner 1990, Flebbe 1993, Welsh et al. 1995, Keleher and Rahel 1996, Welsh et al. 1998). At the watershed scale, elevation, latitude, aspect, and stream size interact to determine annual and seasonal temperature cycles. Thus, the physical location of a stream within the river network influences population life-history characteristics such as spawn timing (Gresswell et al. 1997), growth rate (Lobón-Cervía and Rincón 1998, Campbell 1999), and the timing of smolt migration (Whalen et al. 1999a). At smaller reach or segment scales, the type and density of riparian vegetation and the degree of groundwater input influences the stability of seasonal and diel temperatures (Smith and Lavis 1975, Gregory et al. 1991, Allan 1995). These reach-level patterns can influence behavior (Fraser et al. 1993), food digestion and assimilation (Cunjak and Power 1987, Cunjak et al. 1987), ability to hold position against water current (Rimmer et al. 1985, Graham et al. 1996), and inter- and intra-specific survival and distribution (Torgersen et al. 1999, Harvey et al. 2002). Finally, at the habitat unit scale, cool water inputs from tributaries, intergravel flow through river bars, and streamside subsurface sources can thermally stratify individual pools, providing cool water refuges for individuals of multiple age classes (Nielsen et al. 1994). Thus, species

assemblages and distributions are structured by a combination of larger scale geomorphic and climatic conditions, as well as biotic and abiotic conditions of the local environment. Therefore, biological and physical conditions at any site should be viewed in the context of the larger geologic, climatic, and geomorphologic conditions of the system as a whole—a multi-scale approach.

Transferability Across Scales

Smaller scale studies generally focus on identifying physical features used by individuals (Bustard and Narver 1975, Cunjak 1988, Nakamoto 1994), how this habitat use changes ontogenetically and temporally (Rimmer et al. 1983, 1984, Baltz et al. 1991, Heggenes et al. 1993, Whalen et al. 1999b), and how these habitat preferences differ by species (Bisson et al. 1988, Fausch 1993, Heggenes et al. 2002). These types of studies have also identified bottlenecks limiting the production of different salmonids (McMahon and Hartman 1989, Tschaplinski and Hartman 1983, Solazzi et al. 2000), and have contributed to our understanding of both intra-specific (Symons and Heland 1978, Kennedy and Strange 1986, Harvey and Nakamoto 1997) and inter-specific (Hearn and Kynard 1986) competition. Smaller scale studies often generate models based on correlations between habitat use or availability and fish abundance, incorporating the concepts of optimal and suitable habitat. However, individuals are not always found in suitable habitat (Bozek and Rahel 1991) and not all recruits are necessarily produced from those habitats with the highest densities (Grossman et al. 1995). Moreover, some preferences (e.g., nose velocities) are transferable within systems (DeGraaf and Bain 1986, Morantz et al. 1987), whereas others (e.g., substrate and depth) vary across systems and partly depend on habitat availability (Bozek and Rahel 1992). These habitat-based models may have low predictive power across large areas because 1) biological factors such as abundance of prey,

predators, or competitors are sometimes excluded (Grossman et al. 1995), 2) specific mechanism(s) responsible for the selection (and the subsequent consequences on individual fitness) are not identified (Grossman et al. 1995), 3) stream level variability may not be incorporated (Dunham and Vinyard 1997), and 4) frequent, stochastic disturbances common at smaller scales make these systems less predictable (Levin 1992).

Large-scale investigations generally address the shaping of aquatic systems and their inhabitants by climate variation (Meisner 1990, Keleher and Rahel 1996), geology and geomorphology (Platts 1979, Lanka et al. 1987, Nelson et al. 1992, Richards et al. 1996, Kruse et al. 1997), and land use and land cover (Connolly and Hall 1999, Bradford and Irvine 2000, Waite and Carpenter 2000, Paulsen and Fisher 2001). Examining larger spatial scales over longer time frames produces more generalized models; however, detail is sacrificed (Levin 1992). Such models can link species presence, absence, or composition to stream or watershed characteristics (e.g., stream size, geology, climate, land use) but often with mixed results. The omission of natural temporal variation in population abundance can obscure relationships (House 1995, Bradford et al. 1997), and study designs with limited spatial coverage (Baxter et al. 1999, Rieman and McIntyre 1996, Pess et al. 2002) or short sampling periods (Rieman and Myers 1997) may not reliably indicate population trends or associations with habitat or habitat change. Also, many larger scale analyses synthesize small-scale datasets created for other purposes. Merging of such data sets often combines data with inconsistent sampling frequencies, efficiencies, and representativeness, and analyses can produce unreliable or inexplicable results (Rieman et al. 1999).

In sum, key information can be obtained at any scale of study. However, combining this knowledge across scales and disciplines has proven difficult (Levin 1992, Imhof et al. 1996). It

is generally accepted that occupancy of a given habitat is determined by a combination of small-scale biotic and abiotic conditions experienced by an individual, within the constraints set by the larger landscape. As a result, there has recently been a rise in multi-scale investigations that seek to understand the larger scale variables structuring aquatic species and, within these larger variables, the specific habitat characteristic influencing populations (e.g., Watson and Hillman 1997, Baxter and Hauer 2000, Labbe and Fausch 2000, Pess et al. 2002). As this research continues, recovery planning should emphasize maintaining a high degree of habitat heterogeneity to ensure the availability of sufficient habitat combinations to sustain multiple populations and species (Ward 1998). The focus should be on maintaining proper ecosystem function rather than on managing for specific habitat criteria. This focus requires a thorough understanding of the linkages between biological and physical processes within and across scales (Lewis et al. 1996).

Scale in Recovery Planning

Anthropogenic activities can limit viability and persistence of salmon populations by affecting the quantity or quality of stream, estuary, and nearshore habitats. For example, forestry practices can alter the volume and timing of runoff or sediment delivered to streams, and can reduce the volume of large woody debris, number of off-channel habitats, streambank stability, channel roughness, and water quality (Meehan 1991). Agricultural practices often reduce riparian vegetation and increase streambank instability, sedimentation, hydromodification, and levels of nutrients and pesticides (Waters 1995, Waite and Carpenter 2000). Urbanized areas experience increased sedimentation and pollution, along with many of the problems associated with agriculture (Waters 1995, Waite and Carpenter 2000). Dams and other forms of hydromodification can alter natural flow regimes, isolating river channels from their floodplains

and riparian systems and altering the natural processes of sediment erosion and deposition (Poff et al. 1997).

Individually and in combination, these anthropogenic impacts alter salmonid abundance and survival at various life stages, and ultimately influence the maintenance and recovery of populations and ESUs. Anthropogenic activities generally reduce habitat and community complexity (Ward 1998, Gorman and Karr 1978), and species with more rigid habitat requirements are more susceptible to habitat degradation and displacement by other species (Nelson et al. 1992). Therefore, understanding how land use activities alter natural processes and conditions within stream systems is critical for recovery efforts. The following subsections will discuss how scale can be incorporated into assessments of habitat factors that prevent populations or their parent ESUs from meeting the four categories of recovery goals: population abundance, population growth rate, spatial structure, and diversity (McElhany et al. 2000).

Population Abundance

The extinction risk faced by a population is inversely related to abundance, making it possible to use abundance to define broad risk categories (McElhany et al. 2000). To address how habitat changes might have altered fish abundance (and thus population risk), it is necessary to determine how abundance changes with land use. Historical and recent trends in abundance can be identified through such measures as redd counts, dam and weir counts, spawner and carcass surveys, harvest estimates, and juvenile counts. Once population trends have been identified, they can be compared to trends in land use activities for possible correlations.

For example, Nehlsen et al. (1991) examined anadromous fish stocks in the Pacific Northwest and California to identify stocks with high or moderate risk of extinction. Recent escapement trends for seven anadromous salmonid species were utilized to assess stock risk.

One finding was that the native upriver fall chinook salmon (*Oncorhynchus tshawytscha*) population in the upper Columbia was strong within the Hanford Reach, Washington (Huntington et al. 1996 also classify this as the farthest inland population in healthy condition), while native, naturally spawning populations had declined to very low levels within the Snake River. Using fish passage counts at individual dams, Dauble and Watson (1997) estimated that fall chinook salmon spawning in the Hanford Reach increased from 60% of the total run above McNary Dam in the 1960s to 80% of the run in late 80s and early 90s. In contrast, the proportion of the run entering the Snake River declined over this period from 40% to less than 5%. This decline in the mid 1960s and the 1970s was attributed to losses of juveniles passing through turbines and delays of migrations in Snake River reservoirs (Raymond 1979).

Once these population trends were identified, it was possible to investigate how they might have been shaped by habitat changes. Dauble and Geist (2000) examined the spawning habitat characteristics of the Hanford and Hells Canyon (Snake River) sites to assess how hydroelectric development had influenced spawning habitat availability. Redds were found across a greater range of depths and dominant substrate sizes in the Hanford Reach than in Hells Canyon. They concluded that the Hanford Reach population has remained viable largely due to a geologic template that is highly compatible with their life history requirements. In contrast, the Hells Canyon population must contend with poor habitat quality and quantity coupled with the elimination of upstream and downstream populations through migration blockage and habitat inundation associated with dam construction.

The focus of abundance examinations is to identify spatial or temporal trends in abundance and to correlate these trends with landscape features or land use activity across or within watersheds. However, the resulting correlations cannot identify causation. Rather, they

highlight more detailed investigations needed to uncover impaired processes and identify necessary recovery efforts.

Population Growth Rate

Investigations of population growth rate over broad spatial scales often involve identifying gross differences in the growth rate of multiple stocks and the role of climatic conditions or marine and freshwater processes in shaping these patterns. For example, Mueter et al. (2002) examined the effects of ocean temperature on survival rates of pink salmon (*O. gorbuscha*), chum salmon (*O. keta*), and sockeye salmon (*O. nerka*) from Washington to Alaska. The growth rate was quantified for each of the 120 wild stocks by computing a survival rate ($\log(R/S)$) from eggs to adult recruits (R) after accounting for density-dependent effects of spawner abundance (S). Temperature effects were estimated for each stock separately using a generalized Ricker model then the distributions of parameters from these single stock models were examined to identify geographic differences across stocks. The results suggested that temperature effects on survival rates are consistent within regions, but that northern and southern stocks respond in opposite ways to temperature variations. Similarly, Peterman et al. (1998) used a multi-stock approach to examine spatial and temporal characteristics of environmental processes influencing the growth rate of 29 sockeye salmon stocks from British Columbia and Alaska. Again, population growth rate varied at the regional scale, with survival rates of Fraser River stocks being influenced by large, stock-specific interannual variability, while Bristol Bay stocks were driven by stronger regional-scale processes acting at both interannual and decadal time scales. Thus, these larger scale analyses tend to identify growth rate trends for multiple stocks spanning wide areas, but cannot link these patterns to specific causal factors.

Focusing investigations on specific populations can identify how growth rate responds to a specific anthropogenic activity, but again the causal mechanism(s) can only be suggested. Schaller et al. (1999) examined the response of stream-type chinook salmon stocks within the upper (Snake River and upper Columbia River) and lower Columbia River regions to hydropower development. Spawners were estimated for each index area based on expanded ground and aerial redd counts or live fish and carcass counts, and age-structured spawners were expanded into recruitment. For each index stock, spawner and recruit data were classified into two time periods: 1) pre-1970 brood years prior to the completion of the last two Snake River dams and 2) post-1974 brood years marking the initiation of mass transportation of smolts around the Snake River dams and passage improvements. While all three regions showed a general decline in growth rate and survival between the two time periods, the declines in upriver stocks (which were most affected by hydropower development) were more severe. Thus, differences in the growth rate and survival rates between upriver and downriver stocks coincided in space and time with the development and operation of the hydropower system. However, more detailed analyses are needed to examine potential causal mechanisms for these findings (described in the previous “Population Abundance” subsection).

Analyses of stage-specific growth rate (i.e., realized over some discrete portion of the life cycle) can also be an important information source, particularly where the dynamics of one life stage dominate the dynamics of the entire life cycle (McElhany et al. 2000). For example, Nickelson et al. (1992) estimate coho salmon (*O. kisutch*) smolt production for coastal Oregon basins using juvenile density estimates by habitat type for different seasons. Fully seeded streams were sampled each season and habitat was classified using a modified version of the habitat classification scheme described by Bisson et al. (1982). Results indicated that production

of smolts in Oregon coastal streams can be limited by the availability of winter habitat. Once such bottlenecks are identified, they can guide the recovery planning process toward measuring the quantity of critical habitats available in non-impacted versus impacted sites. Also, historical reconstructions can be conducted to quantify the actual losses of these critical habitats in impacted areas. For example, Beechie et al. (1994) estimated the magnitude of lost rearing habitat and the subsequent loss in coho salmon smolt production by habitat type and form of impact within the Skagit River basin. Using a combination of field surveys, maps, and orthophotos to estimate current and historic habitat, as well as survival to smoltification rates from Reeves et al. (1989), they found a 34% decrease in smolt production capacity of winter rearing habitats from historical production. Hydromodification, largely due to diking, accounted for 91% of the total smolt production losses for winter rearing areas. This type of information is useful in the recovery process, for it can highlight the processes that need to be restored to improve the health of the local habitat, as well as set recovery priorities based on the degree of degradation. See the “Habitat Analyses for Phase II Recovery Planning: Identifying Ecosystem Restoration Actions” section, page XXX, and the “Prioritizing Potential Restoration Actions within Watersheds” section, page XXX, for related discussions.

Spatial Structure

The spatial structure of a population refers to the spatial distribution of individuals within a population and the processes generating that distribution. It is dependent on habitat quality, spatial configuration, and the dynamics and dispersal characteristics of individuals in the population (McElhany et al. 2000). The heterogeneous stream environment may be viewed as a series of habitat patches at an array of spatial scales. The likelihood of individuals inhabiting each patch is dependent on the quality of habitat in the patch as well as the ability of individuals

to move between patches. Therefore, a proper investigation into how habitat changes might have altered the spatial structure of a population requires understanding the small- and large-scale influences on both habitat patch dynamics and salmonid movement.

Comparing historical presence of a species to predicted or current occurrence can help describe the distribution of a species and populations across a landscape, their degree of isolation, and their size. Rieman et al. (1997) looked at potential historic and known distribution of bull trout (*Salvelinus confluentus*) within the interior Columbia River basin, representing 20% of the species' global range. Available information on the presence and status of bull trout was summarized and validated, and classification trees were developed to predict bull trout presence and status using landscape features and management history. Bull trout were widely distributed within their potential range, occurring in about 36% of the subwatersheds; however, much of this distribution was patchy and distributions disjunct. Bull trout were estimated to occur in 44% and be strong in 12% of the historic spawning and rearing subwatersheds. Such coarse examinations can highlight the distribution of populations across the landscape and give some indication of population strength based on size. More detailed work is needed to address patch size and dynamics within individual systems.

Dunham and Rieman (1999) examined 81 patches ($\geq 10^3$ m) within the Boise River to identify patterns in juvenile bull trout occurrence. Habitat patches were defined as stream catchments above 1,600 m elevation with an accessible perennial stream (Rieman and McIntyre 1995), and patches, road densities, and interpatch distances were estimated using GIS methodologies. Trout occurrence, stream width, and gradient data were obtained in the field. Results indicated that the large-scale geometry of catchments can strongly influence the distribution of aquatic species, although smaller scale factors also affect distribution. They

concluded that bull trout populations in larger, less isolated, less disturbed patches are more likely to persist and that it is critical for disturbance within these habitats to be minimal. On the other hand, small, isolated, disturbed populations and habitats are at risk. Thus they speculate that conservation and restoration opportunities might provide best results if centered within those patches of intermediate size or isolation. In a comparison of patch size distributions for Lahontan cutthroat trout (*O. clarki henshawi*) and this bull trout data, Dunham et al. (2002) found that the size distributions were similar between the two species and skewed toward smaller patches, so that very few large patches may be critical to each species. Both species are likely to occur when patch size exceeds 10^4 ha in area, and bull trout are more likely to occur in smaller patches that may in turn explain their occurrence in a large percentage of suitable habitat.

Identifying how species are distributed across the landscape and utilize habitat patches is a necessary first step in the recovery planning process. Linking this information to their dispersal and migratory behavior can highlight “stepping stone” patches: critical patches connecting large habitat areas. Once identification of critical habitat patches is achieved, it is possible to examine how land use actions have reduced or eliminated connectivity of these patches as well as altered or destroyed the patches themselves. This information can then dictate the scale of recovery efforts by pinpointing the processes that are in need of restoration and by identifying local, specific recovery actions.

Diversity

Diversity refers to the distribution of traits within and among populations (McElhany et al. 2000). Genotypic and phenotypic diversity occur at all scales; however, the majority of genotypic diversity is contained within stocks while most phenotypic diversity is greater across populations and landscapes (Healey and Prince 1995). Thus successful conservation must focus

as much on ensuring habitat quality and connectivity as on genotypes (Healey and Prince 1995). The genetic controls are beyond the scope of this document, but it is possible to examine how habitat changes influence the phenotypic expression of traits. Differences in these traits can have adaptive value and should be maintained, even if they don't have a genetic basis, and can be expressed as changes in morphology, fecundity, run timing, spawn timing, behavior, smolt age, age at maturity, and egg size (McElhany et al. 2000). For example, in Carnation Creek, British Columbia, a decline in 0+ coho salmon densities and an accompanying increased growth rate following logging resulted in greater rates of 1+ smolt production (Hartman et al. 1996). This shift in the age and size composition of smolts resulted in an increase in the variability of adult production (Holtby and Scrivener 1989). Unfortunately, long-term databases that can illustrate the results of anthropogenic activities are rare. Thus other ways to determine widespread threats to diversity are needed.

One approach is to conduct a regional examination of changes in species diversity under the assumption that where species diversity is decreasing, phenotypic and genetic diversity might also be decreasing. For example, Frissell (1993) expanded on the identification of Pacific salmon stocks at risk synthesized by Nehlsen et al. (1991) to map region-wide patterns in fish diversity. He used data compiled for inland fishes (Williams et al. 1989) and anadromous stocks (Nehlsen et al. 1991), focusing on the Pacific Northwest and California, which highlighted species and stocks at risk. Mapping units were based on a drainage basin size of 50 to 2,000 km² and the basins categorized according to the number of species (0-1, 2-3, 4-5, or 6-8) classified as extinct, endangered, or threatened. Isopleths were then fit between the categories. Results indicated a general increase in endangerment from north to south. Regions with 4+ species at risk were of particular concern and species declines were attributed to large-scale dams and

irrigation projects. These types of regional examinations can identify locations where diversity is declining, and consequently where more detailed examinations are needed to identify processes responsible for the decline, as well as efforts needed to remedy the problem.

A variety of bioassessment tools can also help identify where anthropogenic activities have eroded stream health and threaten diversity. One method is to assess habitat degradation using invertebrate species assemblages. For example, the reference-condition approach requires the development of a reference database containing invertebrate assemblages and matching habitat descriptors for a large number of minimally disturbed reference sites. Invertebrate assemblages at these reference sites are described, classified, and related to habitat attributes to develop predictive models. The resulting reference models can then be compared to test sites to identify impairment. Results will highlight impaired areas, but will not specify the underlying causes. See the “Habitat Analyses for Phase II Recovery Planning: Identifying Ecosystem Restoration Actions” section, page XXX, for a more detailed discussion of these methodologies.

Another method of examining the effects of land use on stream health and diversity is to characterize changes in species assemblages across land use gradients. In affiliation with the U.S. Geological Survey’s National Water-Quality Assessment Program, Waite and Carpenter (2000) utilized this approach to identify how natural and land use gradients influence biological assemblages within the Willamette River basin in Oregon. Both field data (fish sampling, water quality sampling, and habitat measurements) and GIS data (land use, drainage area, and elevation) were used, covering reaches within seven major subbasins and three ecoregions. They found physical habitat a better descriptor of fish assemblages across ecoregions (i.e., all sites), whereas water chemistry was better within the Willamette Valley ecoregion. They also suggested that the reduced riparian quality and increased water temperatures, nutrient, and

sediment supply found in small agricultural and urban streams cause fish assemblages to shift from those dominated by native, intolerant species to those dominated by introduced or tolerant species. Amount of riffle habitat, the quality of the riparian cover, and the maximum water temperature were the overriding variables describing variation among land uses. These biological assessments provide an opportunity to monitor long-term changes in community composition or stream health (Rabeni 1992) at multiple scales, highlighting areas of possible impairment due to land use activities, and identify possible causes that can then be further investigated.

Putting It All Together

The foregoing was intended to illustrate how the effects of habitat change on the four components of population viability can be examined over multiple scales. However, the components are not exclusive and information gleaned from one component is often useful for other components as well. This section concludes by illustrating in a general sense how to put all the components together, incorporating scale.

The first step is to identify patterns by examining how abundance or distribution changes over time and space. Results of these examinations can often be applied to several of the recovery goals. For example, the work done by Rieman et al. (1997) in the Columbia and Klamath basins and by Dunham and Rieman (1999) in the Boise River looked at distributions of bull trout across the landscape, and in the former, predicted historic distributions. Each study was able to identify increased patchiness of bull trout populations over time, as well as relative population sizes. Each study also examined the spatial isolation of populations to estimate their relative strength, diversity, and degrees of risk. Similarly, Thurow et al. (1997) examined the distribution of other native salmonids within the interior Columbia River and portions of the

Klamath and Great Basins to determine their distribution and status. Their work highlighted the proportion of the potential range currently occupied by each species and the relative strength of the populations. They found all taxa to have narrower distributions, fewer areas with high diversity, and lower percentages of strongholds than in their estimated potential historical conditions. Strongholds were generally found to be rare and not well distributed across the landscape. Thus, examining abundance and distribution across space and time can illustrate areas retaining historic diversity and ecological structure as well as those that have possibly been altered by human influence.

The large-scale studies described above broadly correlate landscape factors (land use, climate, other species) to population trends, and highlight systems that warrant closer examination. Examination of finer scale components of a system (i.e., at watershed or smaller scales) can then identify causation (e.g., road density, hydromodification, water quality degradation), which helps understand how specific ecological processes have been impaired and indicates potential pathways for restoring them. The scale of the analysis constrains interpretations of results (Wiens 1986), and increasingly fine scales of assessment obtain more detailed information. For example, Nehlsen et al. (1991) noted a decline in summer steelhead trout (*O. mykiss*) within most river systems in California. They identified the Eel River population to be at moderate risk of extinction and mentioned how floods in 1964 severely affected populations throughout California due to extensive erosion and habitat damage in watersheds stressed by poor land management. Harvey et al. (2002) highlighted potential mechanisms contributing to these declines, speculating that habitat changes caused by the flood altered summer temperature regimes and allowed extensive invasion by the non-native Sacramento pikeminnow (*Ptychocheilus grandis*) decades later. The investigators concluded

that 1) restoration of riparian vegetation within the watershed could reduce the range and ecological impact of the pikeminnow, and 2) increased riparian vegetation and improved hillslope conditions could enhance native salmonid habitat by moderating the influence of peak flows and sediment supply on channel stability, thereby improving thermal regimes and instream habitat. Thus, results of larger scale work identify a problem, while smaller scale work highlights specific anthropogenic activities and ecological impairment underlying the problem and, consequently, more specific conservation and restoration strategies.

As aquatic ecosystems are arranged in an interconnected array of hierarchical systems, any study of their processes and inhabitants can be organized in a similar hierarchical manner. Species patterns and relations to habitat and anthropogenic activities can be seen at any scale; however, the scale examined dictates the level of detail that can be inferred from the results and any subsequent interpretations. By examining the abundance and distribution of species and associating these with land use activity at multiple scales, patterns in the spatial structuring and diversity of populations can be identified and thus their level of risk. Such information can then be used at fine scales to identify the ecological processes impaired and to prioritize conservation and restoration activities.

APPENDIX B. ESTIMATING CHINOOK SALMON SPAWNER CAPACITY OF THE STILLAGUAMISH RIVER

One important task of TRTs is evaluating both current and historical capacity of the habitat to support salmonids as juveniles and adults. These estimates are needed to evaluate whether population goals derived from population viability analyses could be supported by currently or historically available habitat. For example, if viability estimates indicate that 5,000 fish are needed for a population to be viable, estimates of spawner capacity can indicate whether 5,000 fish is within the bounds of what the watershed may have historically supported.

Riverine habitats function at different spatial and temporal scales, ranging from the watershed level down to microhabitats (Frissell et al. 1986). Larger scale systems (such as watershed and segment) generally operate over a 100 to 1,000 m linear spatial area and persist for 1,000 to 1 million years (Frissell et al. 1986). Extrinsic forces driving these larger systems include glaciation, volcanism, tectonic uplift, climatic shifts, earthquakes, and alluvial or colluvial valley infilling. Reach-scale systems operate at intermediate scales of about 100 m and 10 to 100 years, and are driven by events such as debris torrents, landslides, and log input or washout. Smaller scale, habitat-unit systems are typically controlled by events or processes occurring at spatial scales of 1 to 100 m and over shorter time periods (persistence of 1-10 years) (Frissell et al. 1986). Habitats at this level include pools and riffles as well as glides, rapids, side channels and backwater pools, and have characteristic bed topography, water surface slope, depth, and velocity patterns (Bisson et al. 1982). These habitat units are driven by delivery or routing of sediment, wood, boulders, etc., small bank failures, flood scour or deposition, thalweg shifts, and numerous human activities (Frissell et al. 1986).

As a case study, we estimated the historical and current capacity of the Stillaguamish River for adult chinook salmon (*Oncorhynchus tshawytscha*) based on habitat data at the unit scale (e.g., pool, riffle, and glide). Detailed estimates of salmonid abundance at the habitat-unit scale were extrapolated to the watershed scale to estimate change in total salmonid abundance for the basin. This analysis of capacity allowed us to estimate the maximum number of adult chinook that the Stillaguamish River produced historically, as well as the system's potential for production today.

Methods

Our approach for estimating capacity essentially had two steps. First, we assessed the amount of habitat available for chinook spawning, and second, we associated fish numbers with that habitat. To assess available habitat, we first mapped where adult chinook spawn in the watershed. Within that area, we identified streams either as large main stems or small main stems and tributaries. We quantified habitat area by estimating the total stream length for small main stems and tributaries, and the total stream area for large main stems. Large main stems were further characterized by estimating the proportion of reach-scale habitat that was pool, riffle, or glide, and by how much of each habitat unit was suitable for spawning.

In the second step, we assigned fish numbers to habitats. For small main stems and tributaries, we estimated the total number of redds possible by multiplying stream length by an estimate for the expected number of redds per kilometer. For large main stems, we calculated how many redds would fit into our estimated stream area, by assuming an estimate for redd area. Then we converted estimates of total redds for each reach-scale habitat type to total adults using an estimate for number of adults per redd.

Watershed-scale Chinook Salmon Spawning Distribution

Streams were identified as accessible to chinook salmon based on the location of barriers and stream gradient. The historical distribution of chinook included areas below natural barriers (i.e. waterfalls, cascades) and with streams of low (less than 4%) gradient (Montgomery et al. 1999). Similarly, current fish distribution included areas below anthropogenic barriers (primarily culverts) and natural barriers, and with low-gradient streams. At the upstream end, Granite Falls in the South Fork Stillaguamish River was not included in either current or historical capacity estimates. Historically, few chinook were able to ascend Granite Falls. At present, a fish ladder allows chinook to pass upstream, but observed juvenile outmigrant abundance is low (Footnote: Kit Rawson, The Tulalip Tribes, Natural Resources Dept., 7615 Totem Beach Rd., Marysville, WA 98271. Pers. commun., May 2001). We defined the downstream limit of chinook spawning as the upper extent of tidal influence, which occurs at about the confluence of Cook Slough and Stillaguamish River, and Pilchuck Creek with the Stillaguamish River (Figure B-1).

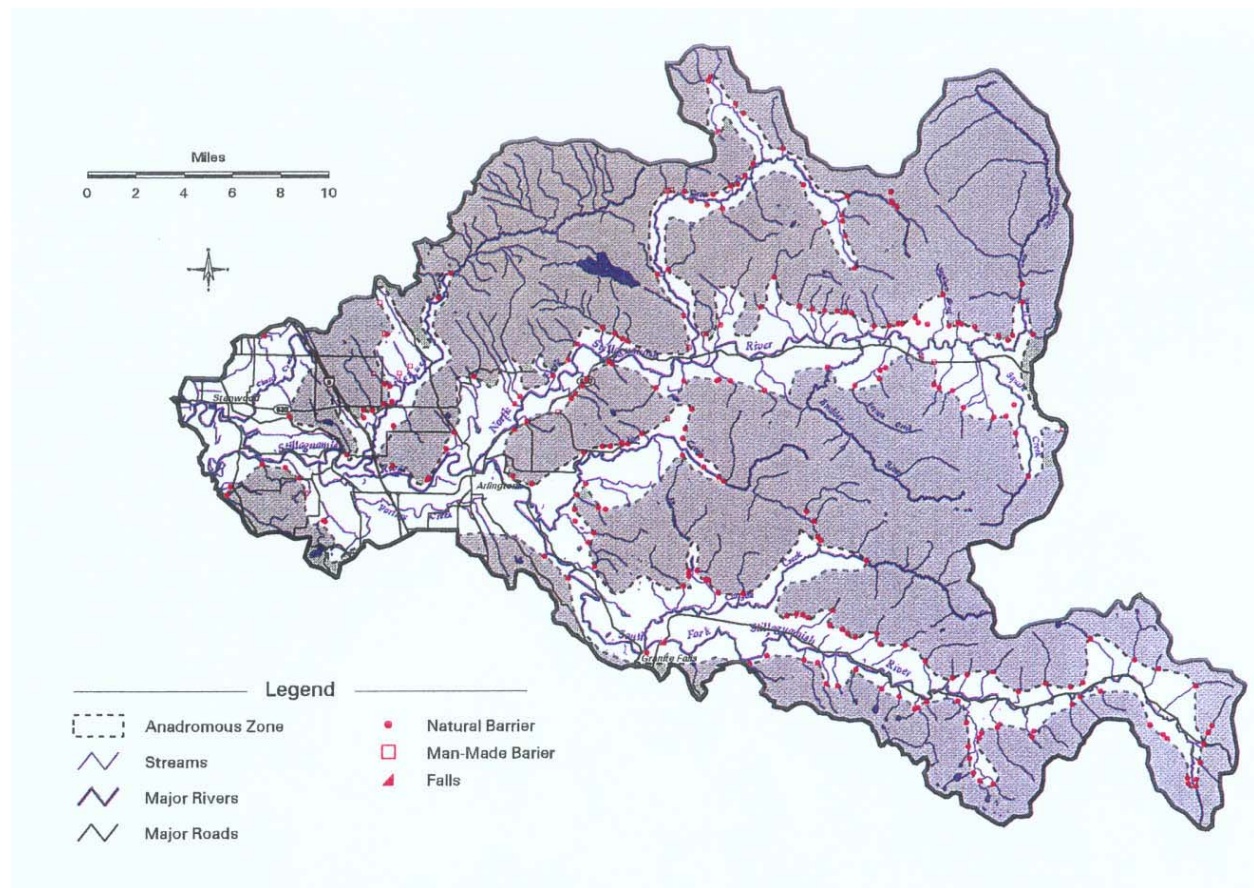


Figure B-1. Basin map of the Stillaguamish River.

Reach-Scale Chinook Salmon Spawning Habitat

The main spawning areas of chinook salmon are in larger tributary streams and main stems of rivers (Miller and Brannon 1982). Chinook use of these freshwater habitats varies depending on the size of bankfull channel widths. Chinook redd density in the Stillaguamish River generally increases up to about 25 m bankfull width, and then sharply declines (Figure B-2) (Montgomery et al. 1999). Similarly, channel morphology varies at different scales as a factor of channel width and large woody debris (LWD) loading. Pool frequency increases with bankfull channel width and LWD loading up to about 20-25 meter channel width, and then drops off (Montgomery et al. 1995). Pools in these smaller streams can be formed by individual pieces of wood, which cross streams and form stable obstructions (Abbe 2000). In larger streams, channel morphology is most significantly affected by large logjams, which form stable obstructions that create pools but at a lower frequency (Abbe 2000). Salmon usually spawn at the transition between pools to riffles, or in areas associated with a lateral bar deposition (Bjornn and Reiser 1991).

Because chinook use of channel types varies with channel width, we classified streams to reach-scale habitat types by bankfull width. Small streams (or mainstem/tributary habitat types) are ≤ 25 m bankfull width, and large mainstem habitats are >25 m width. Additionally, streams less than 5 m bankfull width were regarded as too small for consistent chinook spawner production based on data from the Stillaguamish, Snohomish, Skagit, Suiattle and White Rivers (Footnote: D. Hendrick, WDFW, 111 Sherman St., PO Box 1100, La Connor, WA 98257. Pers. commun., May 2001. J. Doyle, USDA Forest Service, Mt. Baker-Snoqualmie National Forest, 21905 64th Ave. West, Mountlake Terrace, WA 98043. Pers. commun., May 2001). Chinook are occasionally observed spawning in smaller streams (Footnote: D. Hendrick, WDFW, 111

Sherman St., PO Box 1100, La Connor, WA 98257. Pers. commun., May 2001, Vronskiy 1972).

However, we wanted to include stream habitat where the majority of chinook consistently spawn.

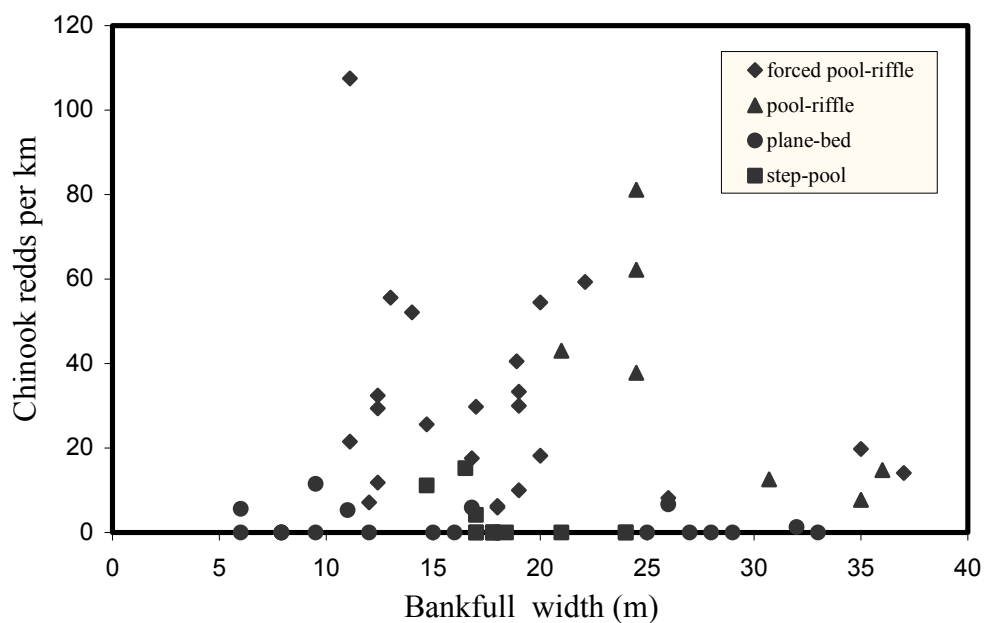


Figure B-2. Comparison of chinook redds per kilometer to bankfull width measurements, Skagit River (based on data in Montgomery et al. 1999).

We estimated bankfull width for both historical and current stream conditions using regression models. For historical conditions, we developed a regression to predict channel width using channel width measurements from 1860s General Land Office Survey notes (Collins and Montgomery 2001) and basin drainage area for the Stillaguamish River:

Historical channel width = $10^{-2.4 + 0.54 * \log(\text{drainage area})}$, Adjusted $R^2 = 0.70$
Basin drainage area was derived using a 30-meter digital elevation model (DEM). For current conditions, a regression model was also developed using channel width data and basin drainage area:

Current channel width = $10^{-1.5 + 0.43 * \log(\text{drainage area})}$, Adjusted $R^2 = 0.68$
Channel widths were measured primarily from aerial photographs, but included field measurements as well.

Using these regression relationships, we developed a GIS-data layer of channel width for the entire Stillaguamish Watershed based on the DEM-derived drainage area data. The channel width predictions were then applied to categorize all streams in a 1:24,000 scale hydrography into the 3 categories: <5m bankfull width, 5 to 25 m width, and >25 m width.

Chinook Salmon Capacity in Small Streams

Habitat in small streams of the Stillaguamish basin could only be described at the reach-scale due to a lack of data describing the proportions of pools, riffles, and other unit-scale habitats. Likewise, the data available for estimating chinook use of small streams (5 to 25 m width) was expressed at the reach scale (i.e., as number of redds per kilometer of stream length Montgomery et al. 1999). Hence, we estimated chinook spawner capacity in small streams as a function of stream length, number of redds per kilometer, and number of adults per redd:

$$N_{\text{adults}} = [(\text{Total stream length} * \text{Redds/km})] * \text{No. Adults/redd}$$

In generating our historical capacity estimate, we calculated the total length of small streams within historical chinook spawning areas from the 1:24,000 hydrography data. Additionally, we included the total length of small, non-pond-like side channels estimated for historical conditions (Pess et al. 1999b). For the current capacity estimate, we excluded those streams and side channels that were blocked by culverts or dams or otherwise isolated (Table B-1).

Table B-1. Estimates of stream length (m) of small streams (5 to 25 m bankfull width) under historical and current conditions for the Stillaguamish River.

Time period	Habitat type	North Fork	South Fork	Main stem
<i>Historical</i>	Small streams and tributaries	209,348	87,148	39,572
	Non-pond-like side channels	10,939	2,667	28,007
<i>Current</i>	Small streams and tributaries	207,173	93,198	45,291
	Non-pond-like side channels	4,462	0	11,079

For estimates of redds per kilometer, we assumed that historically, channel morphology in small streams was determined primarily by LWD loading. Such in-channel obstructions produce forced pool-riffle habitats—channels in which the majority of pool and bar forms are forced by flow convergence, divergence, and turbulent scour associated with obstructions (Montgomery et al. 1995). Forced pool-riffle habitats, formed by woody debris, may be considered indicative of undisturbed conditions (Lunetta et al. 1997). Therefore, data for number of redds per kilometer only include counts from forced pool-riffle habitats (vs. pool-riffle and plane-bed) for historical conditions (Montgomery et al. 1999). Redd density data were collected from 50 reach-level surveys in five tributaries to the Skagit between 1991 and 1996. The median value for redds per kilometer in forced pool-riffle channels was 29.6, and the 10th and 90th percentiles of the range are 7.0 and 57.5 redds per kilometer, respectively (Table B-2).

Table B-2. Values used to vary biological parameters for historical and current capacity estimates.

Percentile	Redd size ^a (m ²)	No. adults per redd ^b	No. redds per kilometer ^c	
			<i>Historical</i>	<i>Current</i>
90th	4.9	3.5	57.5	13.8
Median	14.1	1.9	29.6	2.5
10th	27.9	1.4	7.0	0.0

^a Data from the North Fork Stillaguamish River

^b Data from the Snohomish and Stillaguamish Rivers (D. Hendrick, unpubl. data)

^c Data from the Stillaguamish and Skagit Rivers (Montgomery et al. 1999)

Spawner survey data are also available describing chinook use specific to the North Fork Stillaguamish River from nine river reaches (Table B-3) (Montgomery et al. 1999). However, these data were collected over longer time spans (up to 23 years), and with less certainty that the identified habitats (in particular, forced pool-riffle) persisted over the time period that chinook were observed. These data were useful for our current capacity estimate where redd counts were needed across a range of habitat types, which could vary across time and space (Table B-3). These average counts were first applied to the tributaries from which data were collected (i.e. Boulder River, Squire Creek, etc.). For the remaining small streams, we summarized the data from the nine streams (Table B-3), and applied the median (2.5 redds/km), 10th (0 redds/km) and 90th (13.8 redds/km) percentiles of the range in the current capacity estimate (Table B-2).

We estimated the number of adults per redd using data that describes the number of males per female from: 1) carcass recovery survey data from the North Fork Stillaguamish, Skagit and Snohomish Rivers, 2) Sunset Falls counts from the Snohomish River, 3) broodstock collection data from the North Fork Stillaguamish and Skagit Rivers, 4) mark/recapture study from the North Fork Stillaguamish, and 5) Gill drift net test fishery data from the Skagit River (Hahn et al. 2001, Hendrick unpubl. data). The ratio of males:females plus one is equivalent to

the number of adults per redd. The median value for number of adults per redd was 1.9, and the 10th and 90th percentiles of the range were 1.4 and 3.5, respectively.

To illustrate the full range of potential historic and current capacity values, we calculated capacity estimates using the 10th percentile, median and 90th percentile ranges of all spawner biological variable values (redds per km and adults per redd). Stream length included only one measurement each for historical and current conditions. All capacity calculations include estimates for the North Fork Stillaguamish River chinook salmon population, and the South Fork/mainstem Stillaguamish population.

Table B-3. Data on number of chinook salmon redds for repeated spawner surveys along the North Fork Stillaguamish River (Montgomery et al. 1999).

Location	Years of data	Channel type	Average redd/km
Boulder River	16	Pool-riffle	4.2
Squire Creek	23	Pool-riffle	6.4
Furland Creek	2	Forced pool-riffle	13.8
Ashton Creek	2	Forced pool-riffle	8.8
Browns Creek	10	Forced pool-riffle	2.5
Brooks Creek	4	Plane-bed	0.0
Rollins Creek	6	Plane-bed	0.0
Dicks Creek	4	Plane-bed	0.0
Segelson Creek	2	Plane-bed	0.0

Historical Chinook Salmon Capacity in Large Streams

We estimated historical chinook spawner capacity in large streams (>25 m bankfull width) by determining the amount of area in a river that has habitat suitable for spawning and calculating the number of redds that fit in that area. We then estimated the number of adults as a function of number of fish per redd. The equation for historical capacity in large streams is as follows:

$$N_{\text{adults}} = [(\text{Stream Area}) * (\text{Percent area suitable for spawning})] * \text{No. adults/redd}$$

To calculate stream area in the Stillaguamish Basin, we first generated an estimate of wetted widths across all stream reaches for the time period when chinook spawn (mainly August/September). We measured widths from 1:24,000 digital orthophotos of the Stillaguamish Watershed. The photos were taken in mid- to late-July 1998. Mean monthly flows in July are 25% to 45% higher than in September (generally, the peak of spawning), which means we will overestimate spawner abundance by some fraction. However, available spawning area does not increase in direct proportion to stream flow (i.e. 25% more flow does not equal 25% more spawning area), so overestimation may not be significant. We then developed a regression model to predict wetted widths based on cumulative stream lengths (total stream length above each wetted width measurement) and stream order data, derived from 30-meter DEMs:

$$\text{Wetted width} = 10^{[-2.59 + 0.56\log(\text{Cumulative flow length}) + 0.36(\text{Stream order})]}$$

$$\text{Adjusted } R^2 = 0.77$$

With this regression, we developed a GIS-data layer of wetted width for the Stillaguamish Basin, and associated the wetted width predictions to stream reaches in the hydrography layer. We then calculated stream area for each reach as a function of wetted width and stream length. These stream area estimates were used for calculations of both historical and current capacity (Table B-4). In addition, we included estimates of stream area for non-pond-like side channels for historical conditions (Pess et al. 1999b). All of these side channels are presently isolated from streams accessible to anadromous fish.

Table B-4. Estimates of stream area (m²) under historical and current conditions for the Stillaguamish River.

Time period	Habitat type	North Fork	South Fork	Main stem
<i>Historical</i>	Large streams	2,616,542	1,842,123	1,318,877
	Non-pond-like side channels	<u>70,676</u>	<u>0</u>	<u>98,106</u>
	Totals	2,687,218	1,842,123	2,416,983
<i>Current</i>	Large streams	2,532,976	1,696,890	1,023,863
	Non-pond-like side channels	<u>0</u>	<u>0</u>	<u>0</u>
	Totals	2,532,976	1,696,890	1,023,863

To describe the stream area suitable for spawning under historical conditions, we conducted field surveys in western Washington streams that are in relatively undisturbed condition (North Fork Sauk River, mainstem Sauk River, South Fork Stillaguamish River, Squire Creek, and the South Fork Hoh River). Pess and Abbe (1994) developed criteria for measuring the area suitable for spawning by describing habitat characteristics in the North Fork Stillaguamish River where redds were observed. Geist et al. (2000) similarly used characteristics of spawning habitat measured within local spawning areas to evaluate chinook salmon habitat suitability in the Columbia River. In the Stillaguamish River, substrate diameter averaged 74 mm (range of 45-120 mm) (Pess and Abbe 1994). Depths averaged 0.5 m (range of 0.3-1.5 m), and velocities averaged 0.6 m/s (range of 0.3-1.0 m/s). Healey (1991) summarized comparable values from the literature for water depth and velocity in chinook spawning beds.

In our habitat surveys of reference sites, the average values for substrate and depth were the primary variables used to estimate suitable spawning area, in addition to channel bed morphology. Channel bed morphology indicates the location of subsurface flow (such as at the junction of a pool's tailout and the head of a riffle), which is important in redd site choice by chinook (Vronskiy 1972, Chapman 1943, Russell et al. 1983). Once we identified the location of subsurface flow (i.e. the primary spawning site), we measured the area suitable for spawning

within a habitat (riffle, pool, glide) by surveying the extent of appropriate average substrate and depth values. We calculated historical capacity estimates where percent habitat spawnable was segregated by habitat type (Table B-5). However, we had less certainty that habitat composition was reflective of historical conditions because of the limited size of our reference sites. Therefore, we summarized the percent of area suitable for spawning (“percent habitat spawnable”) across all habitat units combined (Table B-5).

For the capacity estimate, we assumed that redds are uniformly distributed and positioned immediately adjacent to one another, without a larger territorial boundary. Geist et al. (2000) found that when redd densities were near capacity, clusters or redds were uniformly distributed and inter-redd distances ranged from 2 to 5 m. In the Stillaguamish River, redds were estimated to have a median size of 14.1 m² (range of 4.9 m² and 27.9 m² as the 90th and 10th percentiles, respectively) (Table B-2). These redd sizes compare well with redd areas reported by others (Table B-6) (Healey 1991). If we assume redds have an approximately circular shape and thus, radius of 2.11 m, distances between redds in the Stillaguamish River would fall roughly within the range observed by Geist et al. (2000).

We calculated a range of spawner capacity estimates using the 10th percentile, median and 90th percentile of values for all variables (redd size, number of adults/redd, percent habitat spawnable) (Tables B-2 and B-5), except stream area for which we only had one measurement (Table B-4). Our best estimate for historical capacity included percent habitat spawnable with habitat-units combined, but we also calculated capacity estimates with percent spawnable evaluated separately by habitat type.

Table B-5. Percent habitat composition and values used to vary percent of habitat suitable for spawning for historical and current conditions. For current conditions, percent of habitat with observed spawning was calculated using WDFW redd data from 1998, 1999, and 2000, and percent of habitat suitable for spawning was estimated using field measurements of habitat.

Data source	Parameter	Habitat-unit types					Habitat units combined
		Riffle	Glide	Pool	Rapid	Backwater pool	
<i>Historical conditions</i>							
Reference site data	Percent habitat composition	26.0	25.8	47.0	—*	—*	100.0
	Percentile of habitat spawnable						
	90th	20.0	18.0	3.0	—	—	9.5
	Median	17.0	1.0	2.0			3.0
	10th	3.0	0.0	0.0			0.0
<i>Current conditions</i>							
WDFW habitat survey data	Percent habitat composition	46.0	39.0	11.0	3.0	1	100.0
WDFW redd survey data	Percentile of habitat spawned						
	90th	3.4	3.4	3.3	3.8	0	5.5
	Median	1.8	0.2	0.0	0.4	0	0.4
	10th	0.0	0.0	0.0	0.0	0	0.0
NMFS habitat survey data	Percentile of habitat spawnable						
	90th	77.1	50.2	0.0	0.0	0	50.0
	Median	18.8	12.0	0.0	0.0	0	2.7
	10th	1.0	0.0	0.0	0.0	0	0.0

*rapid and backwater pool habitats were not surveyed

Table B-6. Summary of published information of redd size (Healey 1991).

Source	Type*	Redd area (m ²)	
		Range	Mean
Chapman (1943)		2.4–4.0	
Burner (1951)	O	4.8–6.5	
	S	2.4–4.1	
Vronskiy (1972)	S	4.0–15.0	
Neilson and Banford (1983)	S	0.5–27.5	9.5
Chapman et al. (1986)	O	2.1–44.8	17.0

*S = stream-type (spring-run) chinook salmon, O = ocean-type (fall and summer) chinook.

Current Chinook Salmon Capacity in Large Streams

To estimate current capacity in large streams, we used field data for habitat distribution and redd location and field data characterizing habitat suitable for spawning from river miles 15 to 30 of the North Fork Stillaguamish River (Hahn et al. 2001). We digitized WDFW field maps describing habitat units (pool, riffle, glide, etc.) into a GIS database and summarized the habitat composition across the 15-mile reach (Table B-5). We also digitized redd locations for the same 15 mile stretch from 1998, 1999, and 2000 field surveys.

We estimated current capacity similar to methods for historical capacity, except that we first described the habitat composition of the river, and then estimated the percent of habitat suitable for spawning based on those habitat units:

$$N_{\text{adults}} = \frac{[(SA * \%Spwn)_{\text{pools}} + (SA * \%Spwn)_{\text{riffles}} + (SA * \%Spwn)_{\text{glide}}]}{\text{Redd size}} * \text{No. adults/redd}$$

%Spwn = Percent of habitat spawned or suitable for spawning
SA = Stream area

We estimated the percent of habitat that is suitable for spawning in two ways. First, we estimated the actual percent of habitat with observed spawning using WDFW redd survey data. We used GIS to overlay the combined three years of redd survey data with the habitat-unit maps. A buffer was generated around all redd locations to represent area spawned. We assumed a radius of 2.11 m, which would be equivalent to a 14 m² redd size. For each habitat type, the percent of habitat spawned was calculated as the area within buffers divided by the area of the habitat type (WDFW redd survey data in Table B-5). Second, we estimated the percent of habitat suitable for spawning using our field survey data where spawning habitat was characterized by substrate, velocity, depth, and channel bed morphology, as described for historical large stream capacity (habitat survey data in Table B-5). Both estimates of percent habitat spawned and percent habitat spawnable were calculated by habitat type and for all habitat units combined.

Similar to historical estimates for large streams, current spawner capacities in large streams were calculated using the range of values for all variables (redd size, number of adults per redd, percent habitat spawnable by habitat type) (Tables B-2 and B-6), except stream area where only one measurement was possible (Table B-4). Our habitat composition estimates are robust, given the substantial length of stream area surveyed. We therefore considered current capacities with percent spawnable measured by habitat type to be the best estimates. For comparison, we also included capacities where percent spawnable was measured across all habitat units combined.

Sensitivity Analyses

We evaluated the sensitivity of our estimates to parameter values used in the capacity equation, as demonstrated by Gray and Megahan (1981). We conducted the sensitivity analysis

on the historical capacity estimate in the North Fork Stillaguamish River. We selected the range of values as the 10th and 90th percentile of each variable (Tables B-2 and B-5). The base level of capacity was computed using median values for all variables. We then changed each input variable across its lower and upper range of values while holding other variables constant. Finally, the results were plotted for capacity as a relative percentage of change due to variation in each variable (Figures B-3 and B-4).

Results

Spawner Capacity Estimates

Capacity estimates based on 90th percentile of all input variables are as much as three orders of magnitude larger than estimates using the 10th percentiles for both the North Fork Stillaguamish River chinook salmon population and the South Fork/mainstem population, under historical conditions as well as current (Table B-7). The most realistic current capacity estimates are probably those determined using the median values for the biological parameters. It is less clear which statistic is appropriate for estimating percent habitat spawnable under both historical and current conditions. If we were to assume that reference site data underestimate available habitat under historical conditions, we would use the 90th percentile for percent spawnable for historical capacity. This may be a reasonable assumption, given that some of the reference sites (e.g. Upper Sauk and South Fork Stillaguamish Rivers) are not likely as pristine as we expect streams were under historical conditions. For current conditions, we would use the 90th percentile for percent habitat spawnable with the WDFW redd survey data if we were to assume that the North Fork Stillaguamish River is presently underutilized. This assumption may also be fair, given that overall, higher proportions of the habitat types (in particular, riffles and glides)

were characterized as suitable for spawning than were actually observed spawned in by chinook (Table B-5).

For NOAA Fisheries habitat data, the median value for percent spawnable may be more appropriate for capacity estimates. The results from NOAA Fisheries habitat survey data seem high and perhaps overestimate capacity for several reasons. First, data collected on substrate, depth, and flow may not include all the necessary variables that are important for spawning site selection. Other variables that may be important include adjacent or nearby high quality holding habitat (e.g., deep pools with instream cover) and subsurface flow conditions. Second, and perhaps more important, the data used was collected for a different purpose, so the study design originally used for data collection may not suffice to properly estimate capacity for the entire basin.

Using the above assumptions, the best estimate of historical spawner capacity for the North Fork Stillaguamish River chinook salmon population is about 46,800 adults. Current capacity would be in the range of 24,100 to 47,000 adults. For the South Fork/mainstem population, historical capacity would occur at about 50,600 adults, and current capacity would range between 25,000 and 49,700 adults.

Sensitivity analyses

In small streams of the North Fork Stillaguamish River, historical capacity was most sensitive to changes in redds per kilometer, and secondly, adults per redd (Figure B-3). Historical capacity in large streams showed the greatest sensitivity to percent habitat spawnable, followed by redd size (Figure B-4). This estimate was also moderately sensitive to changes adults per redd (Figure B-4).

Conclusions

Estimates of spawner capacity for both small and larger streams vary by several orders of magnitude. Much of the variation is due to: 1) the assumptions and sensitivity around specific variables such as the number of redds per kilometer, adults per redd, percent spawnable area, and redd size, and 2) the data used to develop capacity estimates. Assumptions, combined with a sensitivity analysis, help identify the suite of variables that need additional data collection in the watershed of interest. Each of the variables identified can be measured with existing methods and could be done in one to several field seasons. Developing a database on numbers of redds per kilometer, adults per redd, percent spawnable area, and redd size that are watershed-specific would help identify how the variability fluctuates by ecoregion and population size.

Perhaps a more important conclusion is that utilizing data that was developed for a different purpose and scope may lead to large-scale variations in spawning capacity. For example, the NOAA Fisheries North Fork Stillaguamish dataset was collected in a reach of river that has some of the highest chinook spawning densities in the entire basin. Extrapolating such a dataset to other parts of the basin could lead to an overestimate in spawning capacity, as noted in the preceding example. Developing an adequate sampling design for data extrapolation is an important step that needs to be incorporated prior to data collection and development in order to reduce the amount of variation in spawner capacity estimates.

Table B-7. Range of adult capacity estimates (number of spawners) for the North Fork Stillaguamish chinook salmon population, and the South Fork/mainstem Stillaguamish chinook population, where the combined biological parameter values are varied and percent habitat suitable for spawning is varied. Estimates in bold represent those believed to be the most likely values.

Location	Percent habitat spawnable	Range of biological parameters			Comments
		90th percentile	Median	10th percentile	
North Fork Stillaguamish	Historical conditions				
	Reference site data				Percent habitat spawnable calculated independent of habitat type.
	90th percentile	228,389	46,834	14,465	
	Median	102,484	23,275	5,996	
	10th percentile	45,342	12,582	2,152	
	Current conditions				
	WDFW data				Percent habitat spawnable calculated by habitat type
	90th percentile	129,513	24,141	8,874	
	Median	25,166	4,615	1,855	
	10th percentile	8,533	1,503	736	
	NMFS data				Percent habitat spawnable calculated by habitat type
	90th percentile	1,013,582	189,569	68,340	
	Median	252,106	47,080	17,120	
	10th percentile	16,964	3,080	1,303	
South Fork and mainstem Stillaguamish	Historical conditions				
	Reference site data				Percent habitat spawnable calculated independent of habitat type.
	90th percentile	254,882	50,622	16,503	
	Median	102,182	22,048	6,232	
	10th percentile	32,879	9,080	1,570	
	Current conditions				
	WDFW data				Percent habitat spawnable calculated by habitat type
	90th percentile	137,179	25,027	8,741	
	Median	25,096	4,054	1,202	
	10th percentile	7,231	711	0	
	NMFS data				Percent habitat spawnable calculated by habitat type
	90th percentile	1,086,786	202,720	72,616	
	Median	268,860	49,668	17,598	
	10th percentile	16,286	2,406	609	

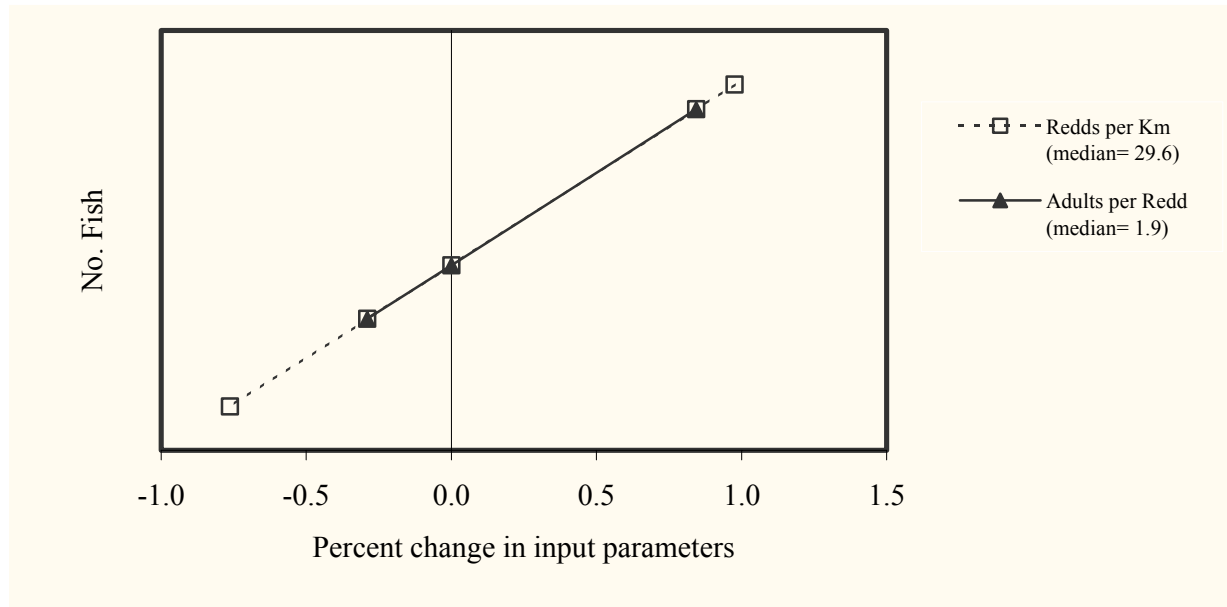


Figure B-3. Range of historical capacity estimates observed by varying parameters in the capacity equation for small streams (≤ 25 m bankfull width) in the North Fork Stillaguamish River.

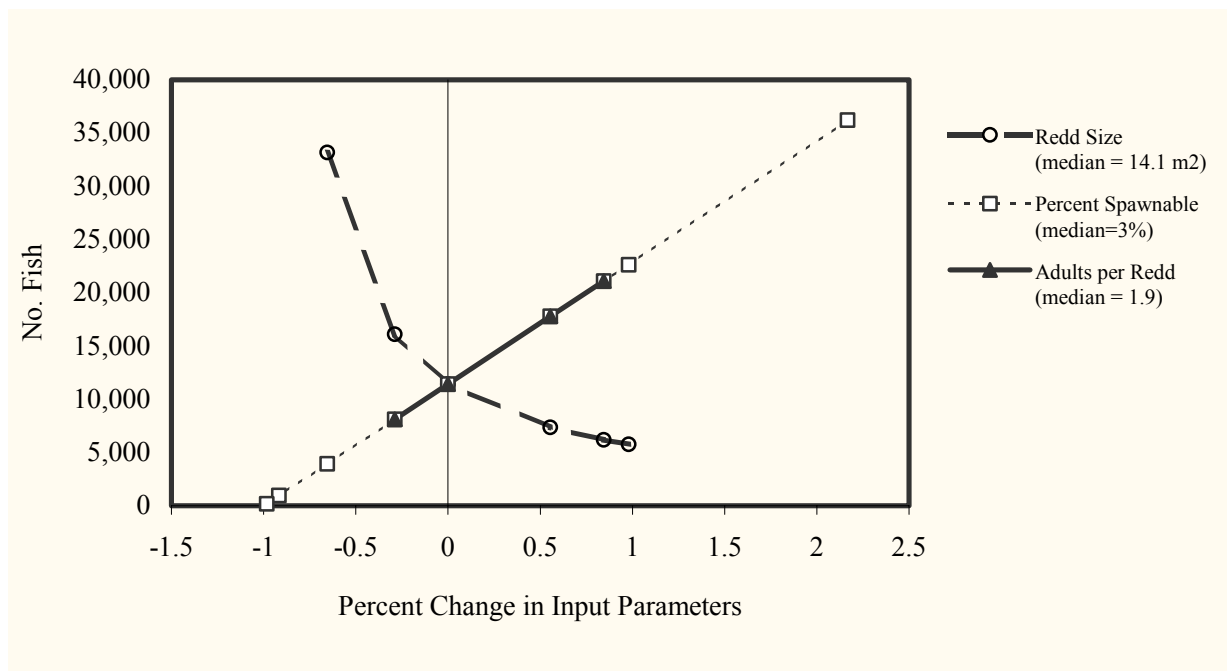


Figure B-4. Range of historical capacity estimates observed by varying parameters in the capacity equation for large streams (> 25 m bankfull width) in the North Fork Stillaguamish River.

APPENDIX C. RESTORATION OF HABITAT-FORMING PROCESSES: AN APPLIED RESTORATION STRATEGY FOR THE SKAGIT RIVER

[Editors' note: This briefly describes the habitat protection and restoration strategy of the Skagit Watershed Council, as well as application of the methods and preliminary results as of 1999. A more detailed description of the strategy and updates of specific results are available on the Skagit Watershed Council website: www.skagitwatershed.org.]

Escapement levels of Pacific Northwest and British Columbia salmon stocks have declined dramatically in the past century due to habitat loss, high levels of harvest, and changes in ocean conditions. Land-use induced freshwater habitat losses were associated with the decline of nearly all of the stocks at-risk in a recent study by Nehlsen et al. (1991). However, the recognition of the causes of these declines and the desire to restore salmon runs has not led to specific plans for recouping habitat losses in large watersheds. Rather, most habitat restoration actions have been conducted in an unplanned and uncoordinated fashion.

In 1997, the Skagit Watershed Council (SWC) was formed to support the voluntary restoration and protection of salmon habitats in the Skagit River basin of Washington State. Today the Council is comprised of 36 member organizations including private industrial and agricultural interests, state and federal agencies, local governments, tribes, and environmental groups. In 1998, the Council adopted a salmon habitat protection and restoration strategy that recognizes the influence of land use and resource management activities on natural landscape processes, which result in changed habitat conditions (SWC 1998). Since 1998, members of the SWC have completed an interim application of the strategy, which identifies causes of degraded habitats in the watershed and restoration actions that are needed to restore habitats over the long

term. In this paper we briefly describe the strategy and methods, and present the preliminary findings of the analyses. We also discuss costs of the analysis relative to projected costs of restoration actions.

Overview of the Restoration Strategy

Figure C-1 is a conceptual diagram illustrating how watershed controls (ultimate and proximate) and natural landscape processes combine to form various habitat conditions. Ultimate controls are independent of land management over the long term (centuries to millennia), act over large areas ($>1 \text{ km}^2$), and shape the range of possible habitat conditions in a watershed (Naiman et al. 1992, Beechie and Bolton 1999). Proximate controls are affected by land management over the short term (\leq decades), act over smaller areas than independent controls, and are partly a function of independent factors (Naiman et al. 1992). Landscape processes are typically measured as rates and characterize what ecosystems or components of ecosystems do. For example, sediment or hydrologic processes in a watershed may be characterized by the rates (volume/area/time period) at which sediment or water is supplied to and transported through specific locations of a watershed. Some riparian related functions can be viewed similarly. For example, large woody debris (LWD) “recruitment” is synonymous with the idea of supply while LWD “depletion” is the result of both LWD transport and decay rates. Natural rates of landscape processes are here defined as those that existed prior to widespread timber harvest, agriculture, or urban development.

The SWC’s habitat protection and restoration strategy describes a scientific framework and set of procedures for identifying and prioritizing activities that restore or protect aquatic habitat (SWC 1998). The scientific framework strives to identify: 1) the natural landscape processes active in a watershed, 2) the effects of land use on natural processes, and 3) the causal

relationships between land use and habitat conditions. It focuses not on the symptoms of watershed degradation but on the fundamental causes, and encourages restoration and protection of natural landscape processes that formed and sustained the habitats to which salmon stocks are adapted. Justification of this approach is based on our understanding from current literature that natural landscape processes create and maintain the “natural” habitat conditions in which native aquatic and riparian species have adapted (e.g., Peterson et al. 1992, Doppelt et al. 1993, Reeves et al. 1995, Ward and Stanford 1995, Beechie et al. 1996, Kauffman et al. 1997).

We apply the strategy by systematically identifying land-use disruptions to landscape processes that form salmon habitat. These processes include peak flow hydrology, sediment supply, riparian functions, channel-floodplain interactions, habitat isolated from salmon access, and water quality. Using a series of diagnostic screens, we locate disturbances to habitat-forming processes, and identify actions (i.e., projects) required to correct the disturbances. This paper reports the progress made by the SWC in applying its strategy within the range of anadromous salmonids of the Skagit River basin in northwest Washington.

Study Area

The Skagit River Basin drains approximately 8,544 km² of the North Cascade Mountains of Washington State and British Columbia (Figure C-2, section A). Elevations in the basin range from sea level to about 3,275 m (10,775 ft) on Mt. Baker. Numerous peaks in the basin exceed 2,500 m in elevation. Average annual rainfall ranges from about 90 cm (35 in) at Mt. Vernon on the lower floodplain, to over 460 cm (180 in) at higher elevations in the vicinity of Glacier Peak. Several vegetation zones occur in the study area. As defined in Franklin and Dyrness (1973), most of the lower elevations are in the western hemlock zone. These forest zones typically include western hemlock (*Tsuga heterophylla*), Douglas fir (*Pseudotsuga menziesii*), western red

cedar (*Thuja plicata*), and sitka spruce (*Picea sitchensis*). Deciduous species in this zone include red alder (*Alnus rubra*), black cottonwood (*Populus trichocarpa*), and big leaf maple (*Acer macrophyllum*). Middle elevations are in the Pacific silver fir (*Abies amabilis*) zone, and higher elevations are in the alpine fir (*A. lasiocarpa*) zone (Franklin and Dyrness 1973).

The Skagit River Basin is comprised primarily of mountain drainages, with fewer lowland subbasins (low topographic relief and low elevation). The hydrographs of most low-elevation forested subbasins are dominated by autumn and winter rainfall floods (Beechie 1992). Conversely, spring snowmelt floods typically dominate the hydrographs of high elevation subbasins in the eastern Skagit. Most areas of the Skagit Basin are of intermediate elevation and exhibit both rainfall and snowmelt floods. Lowland subbasins are generally located in the western valley (rain dominated) and are usually more highly developed by urban and agricultural land use than the forested mountain basins (Lunetta et al. 1997).

Land development (primarily logging and draining or clearing lands for agriculture) began around 1860. About 1590 km² (615 mi², 19%) of the basin is currently in private and State of Washington ownership. Land uses are dominantly agricultural and urban in the lower floodplain and delta areas, and upland areas are generally commercial forests. About 3680 km² (1420 mi², 44%) of the basin lies within the federally owned North Cascades National Park, Mt. Baker and Ross Lake National Recreation Areas, and Glacier Peak Wilderness Area; the U.S.D.A. Forest Service controls an additional 1960 km² (755 mi², 24%) of the basin in the Mt. Baker-Snoqualmie National Forest. Approximately 1040 km² (400 mi², 13%) of the basin is in the Province of British Columbia.

Anadromous salmonids indigenous to the basin include chinook salmon (*Oncorhynchus tshawytscha*), coho salmon (*O. kisutch*), pink salmon (*O. gorbuscha*), chum salmon (*O. keta*),

sockeye salmon (*O. nerka*), steelhead trout (*O. mykiss*), cutthroat trout (*O. clarkii*), and native char (*Salvelinus* sp.). Access to anadromous fishes is generally confined to elevations below 700 m by natural barriers. Upstream migration to the Baker River system has been eliminated by the installation of two hydroelectric dams, but anadromous fish production - primarily coho and sockeye salmon - is maintained through trapping and hauling operations, in addition to the maintenance of sockeye spawning beaches and smolt bypass trapping. The extent of salmon upstream migration in the Skagit River Basin is shown in Figure C-2.

Methods

We analyze disturbances to watershed processes in the Skagit River Basin in two phases. In the first (interim) phase, we locate disturbed habitat-forming processes using a combination of existing Geographic Information System (GIS) data and field-based inventories to identify disturbances to peak flows, sediment supply, riparian functions, channel-floodplain interactions, blockages to salmon migration, and water quality. The second phase (to be largely completed during the next two years) relies solely on field-based inventories. Both phases rely on GIS to analyze and maintain landscape process data over the 8,544 km² area of the Skagit River Basin. This paper describes only the methods and results from Phase 1.

We have used more than 30 different GIS themes and partial field inventories to apply the landscape process screens identified in the Strategy. The existing GIS themes provide low-resolution data covering the entire river basin. These data give us a good overview of habitat-forming processes in the entire basin, but can give erroneous answers to our questions about specific reach level sites (10²-10⁴ meters linear scale). Field inventories provide high-resolution data, but with only limited coverage at present. Because field-inventories are more reliable at

specific sites, the SWC members have made a long-term commitment to collecting field-based information basin-wide.

We analyzed selected landscape processes that form salmonid habitats in the Skagit River Basin. We selected these analyses based on current scientific knowledge of their effects on salmonid habitat and survival of salmon in freshwater, as well as knowledge of how various land use practices affect the processes (Table C-1). We recognize that the list may not include all impacts to salmon in the watershed. However it includes those that are clearly supported by scientific literature and are responsible for a significant proportion of the total loss in salmon production from the basin. For each process we developed a series of diagnostics based on rates derived from scientific literature and local studies. The diagnostics and methods for estimating values are summarized in Table C-1.

We synthesized the ratings for individual landscape processes and functions into a single reach classification that we call the generalized habitat types. The importance of identifying generalized habitat types for watershed restoration is illustrated by Frissell (1993a) and Doppelt et al. (1993), where examples of habitat types are listed along with their biotic objective and restoration tactics. To apply this concept in the Skagit, we derived generalized habitat types based on simple correlations between our understanding of anadromous fish life history strategies and reach level habitat types (approximately 10^2 to 10^4 meters in linear scale) (Table C-2). We assume that relationships between fish life stages and habitat for each indicator species analyzed adequately identifies the “habitats to which salmon stocks are adapted” in an effort to be consistent with our goal.

Our analysis used five species and four life stages to determine generalized habitat types. The life history stages examined were 1) spawning/egg to fry, 2) summer rearing, 3) winter

rearing, and 4) estuary rearing. Several salmonid species present in the Skagit River Basin were excluded from the evaluation due to lack of data or a spatial bias in their distribution not related to geomorphic habitat types. Native char were excluded due to a lack of data describing their habitat preferences over their complete life history. We know the spawning range of native char is bias toward higher elevation headwater tributary basins which is included in the range of historical anadromous fish access (Figure C-1). Cutthroat trout were excluded because of their spatial bias towards the lower elevation rain-dominated subbasins of the Skagit. Coho salmon habitat preference is similar cutthroat, and the coho range includes all of the anadromous cutthroat range in the Skagit, therefore we assume that coho relationships in our analysis adequately represent cutthroat. Sockeye were excluded because the population is limited to one subbasin of the Skagit: the Baker River. While resident rainbow trout (*O. mykiss*) are found throughout the entire river basin, they are assumed to have the same juvenile habitat preferences as steelhead (the anadromous form of *O. mykiss*), which are included in our analysis.

Under pristine habitat conditions (i.e., natural disturbances only) we define reach-level habitat types for anadromous salmonids in the Skagit as either “key” or “secondary” (Table C-2). Key habitat is “critical” for at least one life stage or is a “preferred” habitat type by the majority of life stages considered. Secondary habitat does not provide critical habitat for any life stage and is not a preferred type by the majority of life stages considered. Classification systems described in Hayman et al. (1996), Montgomery and Buffington et al. (1997), Peterson and Reid (1984), and Simenstad (1983) were used to define the different reach level habitat types. Local studies used to designate whether the specific reach level habitat types were “critical”, “key”, or “secondary” for a life history stage included: Beamer and Henderson (1998), Beechie et al. (1994), Congleton (1978), Congleton et al. (1981), Hayman et al. (1996), Montgomery et al.

(1999), and Phillips et al. (1980, 1981). Data from outside the Skagit (Queets River, Washington, in Sedell et al. 1984) were also used to help understand juvenile fish use differences between large main channels and off-channel habitats.

Under disturbed habitat conditions (i.e., both human and natural disturbances) we designated reach-level habitat types as: “key” when all landscape screening results were rated as functioning; “important” when at least one landscape screen is moderately impaired; “degraded” when at least one landscape screen is impaired; “secondary” when channel type is step-pool or steeper; “isolated” when upstream of a man-made barrier to fish migration, or “unknown.” Some reaches are designated as “unknown” because of the high probability of error in rating the riparian condition correctly by land cover types.

Mainstem areas with any of the following conditions were consider degraded: riparian buffer is less than 20 m wide, streambank edge is hardened (e.g., riprap), or levee is present within 60 m of the bankfull channel edge. All other Lower Skagit mainstem areas were considered important. In the estuary, areas that are hydromodified are considered degraded, areas adjacent to levees are considered important, areas at least one distributary channel away from a levee are considered key.

Results and Discussion

Hydrology—Changes in Peak Flow

We estimate 23% of the mountain subbasins in the Skagit are very likely impaired or likely impaired with respect to peak flow hydrology (Figure C-3, section A). In lowland basins, we estimate 7% of the streams historically accessible to anadromous salmon will be impaired when urban and residential areas are fully built out, and 18% will be moderately impaired

(Figure C-3, section B). We use the results shown in Figure C-3 to help evaluate the likelihood of success of proposed restoration projects. In general, we do not support restoration efforts directly in or adjacent to channels that are classified as “impaired” without evidence demonstrating that the proposed work will not fail biologically or physically due to the likely increase to peak flows. We also use the results to identify areas currently in good condition that are planned for future development to an extent that hydrology is likely impaired. For these areas we consider protection actions such as rezoning to a less intensive land use or acquisition. We also identify areas to investigate for potential restoration of hydrologic processes.

Reduced peak flows as a result of flood control change a channel’s ability to create and maintain the suite of diverse floodplain habitats to which aquatic species are adapted (Ward and Stanford 1995). Annual peak flows in the Skagit River Basin have changed since flow regulation through the construction of reservoirs capable of flood storage. Before flood storage capability, floods in the lower Skagit River commonly approached or exceeded 5,500 cms, and floods in water years 1815 and 1856 were estimated at 11,327 and 8,495 cms, respectively. Since the advent of flood storage capability, a flood approaching 5,500 cms has not yet occurred. The number of floods between the 2-year and 100-year return period has been reduced by roughly 50% since dams were built on the Skagit and Baker Rivers (Table C-3). Flood storage on the Skagit has likely impacted channel-floodplain processes in reaches downstream of the dams, but we have not yet quantified the effects. Until we have a better understanding of these impacts, we view the dams as watershed controls (i.e., as shown in Figure C-1). That is, they operate independently of our management control because they are licensed for up to 50 years and are unlikely to be removed. Accepting that this disturbance will not likely be altered during the license period of each dam, the artificial creation of off-channel habitat then may be justified

in stream reaches where off-channel habitat has been lost due to this disturbance. Alternatively, it may be possible to re-establish certain channel-forming flows that have been eliminated in the past.

Sediment Supply

We estimated that 46% of the area in mountain subbasins of the Skagit is impaired with respect to sediment supply (Figure C-4, section A). Our evaluation of the accuracy of the method showed that it correctly estimated the sediment supply rating for seven of the ten subbasins where sediment budget data were available. It over-estimated average sediment supply for two of the ten test basins (i.e., rated them impaired when they are functioning), and under-estimated sediment supply for one subbasin. Therefore, we recognize that this product should not be used to identify potential restoration projects. Rather, it is used for project screening where field-based sediment budgets are not available, and for general planning watershed-level sediment reduction projects. For project screening, project proponents use this map to determine whether the proposed project area is likely to have an impaired (i.e., high) sediment supply. For reaches where sediment supply is impaired, 1) sediment supply in the watershed should be restored to “functioning” levels before downstream reaches are worked on, or 2) evidence demonstrating that the proposed work will not fail due to increased sediment supply should be presented.

Specific sediment reduction projects are identified based on the results of forest road inventories. We focus on forest roads for sediment reduction projects because mass wasting rates from forest roads averaged about 44 times more than mass wasting rates in mature forest (Paulson 1997). Currently, about 1,300 km of road are inventoried with another 3,000 km remaining (Figure C-4, section A). Risk ratings from the current inventory showed that a

significant number of forest roads in the Skagit Basin pose a landslide hazard and potentially threaten fish habitat. Based on this inventory, we will focus initial sediment reduction projects on the high-risk and moderate-risk road segments.

For example, the Bacon Creek watershed has 3.7 km and 18.6 km of high and moderate risk roads respectively (Figure C-4, section B). The high-risk road segments cross relatively more of the landforms sensitive to disturbance for mass wasting (e.g., inner gorges and steeper hillslopes) than moderate or low risk roads. Sediment reduction projects on these roads would reduce the risk of sediment supply increasing due to roads therefore increase the level of watershed protection. Specific road projects primarily involve stabilizing or re-contouring road fills, removing stream crossings or improving drainage conveyance, and improving road surface drainage. For river basin level planning, we consider subbasins with the lowest total cost of road restoration per kilometer of salmon stream as the highest priority.

Riparian Function

Before interpreting the Landsat classification of riparian forests, we used field inventory results from 234 riparian sites to describe the distribution of field-based riparian classifications within each satellite-based forest class (Table C-4). All of the sampled “late-seral forest” sites, and between 92% and 88% of the “mid-seral forest” and “early-seral forest” sites met the > 40 m wide riparian buffer criteria, fitting our functioning designation. Conversely, 90% of the areas mapped as “non-forest” had < 20 m wide riparian buffers, fitting our impaired designation. Areas mapped as “other forest” (ranging from clear-cuts to mature hardwoods) were found to be 43% functioning, 15% moderately impaired, and 42% impaired.

Based on this analysis, we estimate that 29% of the non-mainstem channels in the anadromous zone (by length) are in the non-forest land cover category, and therefore have a very

high likelihood of being impaired and in need of riparian restoration (Figure C-5). Conversely, 19% of the non-mainstem channels in the anadromous zone are in the mid to late seral forest land cover category, and therefore have a high likelihood of being functioning and therefore needing protection. While we cannot accurately map stream reach scale riparian conditions associated with channels adjacent to all GIS land cover types, we can estimate with reasonable accuracy the total of each riparian category at a larger scale. Based on the results in Table C-4, we estimate the percentage of non-mainstem channel length in the Skagit anadromous zone by each land use designation (Figure C-6).

We rely on field inventories to identify actual restoration projects because of the above-mentioned limitations in the satellite classification of riparian forests. We conducted field inventories of riparian forests by walking all streams accessible to anadromous fish and assessing the riparian vegetation conditions for each stream reach. We classified riparian conditions by buffer width, stand type, and age of vegetation within 60 m of stream channels. From these data, we selected all stream segments with forested riparian vegetation less than 40 m wide as requiring planting, and all segments with evidence of livestock access to the stream channel as requiring fencing. Riparian planting and restoration projects have been identified through a series of field inventories. The inventories were completed systematically as four separate projects between 1995 and 1998 in 24% of the Skagit's subbasins (Figure C-5). Together, the inventories identified 130 km of stream corridor for riparian planting and fencing projects.

Isolated Habitats and Disrupted Channel-Floodplain Interactions

The inventory efforts through September 1999 have identified 229 man-made barriers out of 572 channel crossing structures with 32% of the anadromous zone inventoried. In tributary habitat, 143 km of channel is blocked. In the delta, we estimate 185 km (56%) of the channels

have been isolated or lost to salmon access under present conditions (Figure C-7). Isolated channels are those where a topographic channel and water exist, but juvenile or adult salmon access is blocked due to man-made disturbances. Lost channels are those areas that were historically channels, but currently do not have clear a topographic channel or water present.

Because man-made barriers are not evenly distributed throughout the Skagit River Basin and our inventory efforts have focused in areas where barriers are more common, we anticipate that the majority of the isolated habitat in the basin has been found. Based on a subsample of 111 inventoried structures within subbasins of the Skagit with similar land-use intensity as the subbasins yet to be inventoried, we found that 14% of the inventoried structures do not meet fish passage criteria. Therefore, we expect to find about 150 more blockages in non-inventoried areas of the basin, blocking about 60 km (4%) of the estimated length of tributary habitat in the anadromous zone.

Upstream of the Skagit delta, 46 km of stream banks have been riprapped (Figure C-7). In the geomorphic delta, 51 km (62%) of the mainstem channel edge is either hardened, diked within 60 m of the channel's edge, or both. These inventory results provide the basis for identifying potential riprap removal (or modification) projects, primarily where hardened banks no longer protect capital improvements (e.g., house, road).

Generalized Habitat Types

The final result of our analysis is the identification of generalized aquatic habitat types throughout the entire river basin, which are based on salmonid habitat preferences combined with the results of the landscape process screens. The resulting analysis in the Skagit Basin yields a mosaic of reach-level habitat patches (Figure C-8). Key habitat areas have all habitat-forming processes functioning at or near historic levels, and are targeted for habitat protection.

Because protection of habitat is generally considered less expensive than restoration, we view key habitats as some of the highest priority areas for habitat expenditures. Isolated habitats are typically the most cost-effective restoration projects, and therefore receive strong consideration for restoration funding. Important and degraded habitats are both areas that are targeted for restoration.

Secondary habitat will not be targeted for restoration under this strategy. That is, we do not intend to “restore” secondary habitat to key habitat. However, it is important to understand how secondary habitat may function in a watershed in order to protect or restore the other habitat types. For example, the source of degradation may originate in secondary habitat (i.e., the idea of contributing critical areas, discussed in Frissell 1993a). In such cases, restoration of processes originating in secondary habitat areas may be required in order to restore downstream degraded or important habitats.

Identification of Restoration Projects

The main objective of the strategy is to identify habitat protection and restoration projects based on application of the landscape process screens. Together, our analyses have led to the identification of more than 400 individual restoration projects. For example, our analysis of the U. S. Forest Service road inventory identified approximately 650 km of high-hazard and moderate-hazard roads that are candidates for restoration. The total estimated cost for all of these roads (which does not include forest roads on state and private lands) is approximately \$11.6 million. We also identified 122 riparian planting and fencing projects during inventories of only 24% of the river basin, with a total cost estimated at \$1,687,000. Of these riparian projects, 39 are already funded.

We completed migration barrier inventories in 13 out of 38 subbasins, and identified 229 blocking structures. Some blockage removal projects have uncomplicated designs and relatively clear benefits. These projects can each be considered independently of other culverts because salmon currently access the culvert sites, and repair of the structures will provide benefits commensurate with the amount of habitat upstream. By contrast, groups of completely blocking structures on the same watercourse should be considered either in combination or sequentially, and projects that involve flood protection levees or coordination of numerous landowners require feasibility studies to determine suitable restoration actions. Currently we have a list of 36 isolated sloughs and blind tidal channels that require further assessment for design of appropriate solutions.

Each restoration project is mapped on a GIS theme, and relevant data are stored in the associated databases. These themes can be updated as new inventories are completed, or as project status changes (e.g., design phase, construction, completed). Additionally, we can develop related databases for monitoring the effectiveness and costs of different project types. Over time, the GIS maps and databases will help display progress made in restoring habitats in the Skagit River Basin, and will help us modify our actions to more efficiently restore habitat in the basin.

Current Limitations and Future Work

Both lowland and mountain basin GIS-based results give us operating hypotheses for peak flow impairment throughout the river basin. Peak flow ratings for mountain subbasins in the Skagit were developed based on an empirical correlation between land use and elevated peak flows an adjacent basin (North Fork Stillaguamish River) because subbasin flow data are limited in the Skagit. The North Fork Stillaguamish River has exhibited a 38% increase in mean annual

maximum flow between 1928 and 1995 with climatic variables explaining less than 40% of the increase suggesting that changes in the watershed condition has caused the balance of the increase (Beamer and Pess 1999). However, future efforts for the mountain basin methodology must confirm that correlations between land use and peak flows in the North Fork Stillaguamish are a cause-effect relationship, and then identify the appropriate thresholds for land use before re-applying to the Skagit. The lowland basin methodology should be repeated with land cover data that estimates current effective impervious area to complement the results reflecting impervious area at fully developed watershed conditions per zoning designations.

Field-based sediment budgets more accurately estimate the sediment supply in a subbasin, and describe the relative effects of different land uses on sediment supply. Therefore, they will provide more accurate information for project screening and planning than do our current GIS-based estimates. Field-based sediment budgets have been completed for approximately 12% of the total area (Paulson 1997), and sediment budgets for the remainder of the basin will be completed in 2001. In lowland basins, mass wasting is not a dominant sediment supply process, but increased fine sediment supply to channels is directly related to urban, livestock grazing, and agricultural land use. We anticipate future development of a surface erosion and sedimentation screen for these low-slope areas focusing on quantifying surface erosion from agricultural or developed areas.

The U.S. Forest Service is continuing its road inventory. Similar road inventories have not yet been conducted on state or private timberlands. The inventory method appears to be appropriate for identifying segments of road that pose the greatest threat to stream resources. However, it does not identify the types and locations of work needed to reduce the landslide hazard. We anticipate that some inventories will be more detailed than those used by U.S. Forest

Service, and will better identify the specific work actions required for each segment of forest road.

Satellite data do not provide sufficient information for identifying all riparian restoration and protection actions at the stream reach level. The GIS-based riparian screen is reliable for only late and mid-seral conifer dominated forest and non-forest areas. Because of the higher probability of error in rating stream reaches by the remaining land cover types, they are excluded from the interim screen. Therefore, the interim riparian screen is applied to only about 50% of the anadromous zone (based on length). Field inventories are far more reliable than remote sensing data, and can provide sufficient information for stream reach level project planning. Therefore our primary task is to complete field inventory of riparian forest conditions throughout the river basin.

The field-based inventory of man-made blockages to salmon migration has been completed for only a portion of the basin, but the future inventories are fully funded and should be completed by 2001. Areas currently identified as “isolated” are accurately characterized as upstream of man-made blockages to salmon because they are based entirely on field inventory. We assume that some areas yet to be inventoried are “isolated”, although extrapolation from current inventories suggests that no more than 4% of the remaining channel length is likely to be upstream of a man-made blockage to salmon migration. In addition to the remaining blockage inventories, we have yet to complete our inventory of wetland habitat losses in the delta. The wetland inventory should also be completed by 2001.

Water quality parameters such as dissolved oxygen, temperature, turbidity, nutrient loading, and levels of toxic substances are critical to salmon health and survival. Identifying areas where water quality is impaired and the various factors contributing to impairment, and

then addressing the causes of water quality degradation is important to restoring salmon habitat in the basin. Currently, we consider stream reaches, lakes, and estuary areas that are included on the Washington Department of Ecology's Candidate 1998 Section 303(d) Impaired and Threatened Water Bodies listings as "impaired." These water bodies are known to fail Washington State's surface water quality standards, and are not expected to improve within the next two years. We anticipate our future water quality screen to include locations of known point and non-point sources that may contribute to water quality degradation in the basin. These land use indicators will be used to identify areas where water quality problems may exist, and to direct further investigation (e.g., water quality sampling, benthic invertebrate community analyses) to determine if water quality is actually impaired. The continuing objective will be to improve the quality and quantity of water quality data and land use information available to guide restoration and protection of aquatic habitats.

The primary limitations in accurately identifying generalized habitat types are incomplete natural landscape process screens and the accuracy of individual screens used. The consequence of incomplete landscape process screens is an underestimate in the amount of "degraded" and "important" habitat, and an overestimate of the amount of "key" habitat. However, we have high confidence that areas identified as "degraded" are in fact degraded. That is, there is a very low likelihood that areas identified as "degraded" with this analysis will later be identified as "important" or "key" habitat. Conversely, some areas identified as "key" habitat with this analysis will be changed to "degraded" or "important" as more detailed information becomes available.

Conclusions

The SWC first identified its conceptual framework and diagnostic criteria, thus enabling systematic application of a strategy supported by all members. Without this step, a systematic and objective inventory of habitat problems in the Skagit Basin would not have been possible. Following development of the strategy, the Council quickly applied the simplest diagnostic criteria over the entire basin with limited funds. This effort yielded many “no-brainer” project ideas (some are socially “risky”), as well as a good overview of the spatial pattern of disturbance over the entire basin. By contrast, a haphazard inventory or professional judgment system would have produced lists of projects, but would not necessarily give managers the tools to be strategic or comprehensive in restoring salmon habitats. In other words, managers would be unable to focus on important biological hotspots or impaired landscape processes because they lacked a comprehensive understanding of the causes of habitat degradation in the basin.

The strategy recognizes that land use and resource management activities influence natural landscape processes, which result in changed habitat conditions (Figure C-1). Therefore, restoration and protection actions identified by implementing this strategy should be directed at the habitat-forming processes instead of attempting to build specific habitat conditions. Focusing actions on “building” habitat for specific species may be to the detriment of other species and may not be sustainable due to potential conflicts with natural processes (Frissell and Nawa 1992, Kauffman et al. 1997, Beechie and Bolton 1999). Instead, actions implemented by this strategy will aim to create the conditions necessary for natural landscape processes to reestablish at levels similar to those that existed historically which should: 1) result in a high likelihood of long-term project success, 2) protect and restore habitat for all salmonid species as

well as other native aquatic and riparian dependent species, and 3) ensure the effective use of public and private restoration funds.

The SWC overcame diverse interests to develop and apply the interim phase of its restoration and protection strategy in about a two-year period. The field inventory phase of the strategy will be completed during the next two years. The cost of all inventories and analyses required to develop a restoration plan (including a list of required restoration and protection actions) for the 8,500 km² basin is only about \$1.1 million. This total is less than the cost of opening one large isolated estuary channel and wetland complex (\$1.9 million, USACE 1998) or a few culvert blockages on local or state highways (\$250,000 to \$600,000 each—Footnote: D. Brookings, Skagit County Public Works, 1111 Cleveland Ave., Mount Vernon, WA 98273. Pers. commun., Dec. 1997). The estimated cost of potential projects identified during this first phase of applying the strategy is well over \$100 million, suggesting that the total cost of the inventories will be less than 1% of the cost of restoration and protection actions. Moreover, development of the restoration plan should save millions of dollars by avoiding projects that are not effective at restoring salmon habitats. On a per unit area basis, the cost of all interim assessments and final field inventories will total only \$210 per km² assuming that costs remain relatively constant during the next two years.

Application of the strategy gives the SWC the ability to become truly strategic in their restoration and protection efforts by providing a consistent set of principles that guide restoration actions, and by systematically identifying hundreds of restoration and protection projects that can be prioritized and sequenced logically. Having a complete river basin overview of landscape processes and resulting habitat conditions allows the Council to set goals on how much restoration or protection is needed to meet a specific priority. The strategy allows priorities to be

based on locally defined objectives such as recovery of a certain species or completion of certain types of restoration (Lichatowich et al. 1995, Beechie et al. 1996). However, prioritization does not alter the types of projects enacted, but only alters the sequence in which projects are completed (Beechie and Bolton 1999). Currently the SWC prioritizes projects based on the relative cost-effectiveness of different projects, which means that projects protecting or restoring the greatest proportion of anadromous fish habitat function per dollar cost are considered higher priority. Additionally, individual restoration groups may choose projects from any list of projects in order to fulfill their respective missions.

Table C-1. Summary of background and methods for rating individual landscape processes.

Background	Method description	References
<i>Hydrology—peak flow in lowland basins</i>		
The degree of urbanization is correlated with increased flooding, degraded habitat, and lower salmonid populations in lowland basins of Puget Sound. When the 2-year flood magnitude under current land use equaled or exceeded the 10-year flood under “forested” watershed conditions, channels were consistently degraded. Watersheds with effective impervious area (EIA) greater than 10% always had degraded channels whereas watersheds with $EIA \leq 3\%$ had high species and habitat diversity and abundance.	Hydrologic impairment in lowland basins rated based on planned effective impervious area (EIA), which is the weighted average EIA upstream of the stream reach under fully developed conditions per land use zoning designation. Weighted average EIA was calculated using GIS by assigning EIA values to polygons based on land use zoning designations. $EIA \leq 3\%$ is considered “functioning”, EIA between 3% and 10% is “moderately impaired”, and EIA $>10\%$ is “impaired.”	Booth and Jackson 1997, Dinicola 1989, Moscrip and Montgomery 1997, WDF 1994
<i>Hydrology—peak flow in mountain basins</i>		
Increased peak flows result in an increased frequency of channel forming and bed mobilizing flow events, leading to channel destabilization, less complex habitat, and increased bed scour depths significantly affecting salmonids. Two common land use causes of increased peak flow in forested mountain basins relate to hydrologically immature vegetation and forest road drainage. Hydrologically immature vegetation has relatively low canopy density in winter, allowing increased snow accumulation and melt, resulting in higher runoff rates than areas with hydrologically mature vegetation. Forest road ditches extend the channel network, resulting in more rapid routing of water to main stream channels than basins without road networks.	Peak flow ratings for mountain subbasins in the Skagit were developed based on an empirical correlation between land use and elevated peak flow in an adjacent basin because subbasin flow data are limited in the Skagit. Subbasins with more than 50% watershed area in hydrologically immature vegetation due to land-use and more than 2 km of road length per km ² of watershed area are rated “very likely impaired”. Subbasins exceeding one or the other of the criteria are considered “likely impaired”. Subbasins that do not exceed either criterion are considered “functioning.”	Beamer and Pess 1999, Jones and Grant 1996, Lunetta et al. 1997, Montgomery 1993, WFPB 1995

Table C-1 continued. Summary of background and methods for rating individual landscape processes.

Background	Method description	References
<i>Sediment</i>		
Clear-cutting and forest roads increase landsliding and the supply of coarse sediment (>2 mm diameter) to stream channels, although fine sediments (< 2 mm diameter) are also delivered by mass wasting. Large increases in coarse sediment supply tend to fill pools and aggrade the channel, resulting in reduced habitat complexity and reduced rearing capacity for some salmonids. Large increases in total sediment supply to a channel also tend to increase the proportion of fine sediments in the bed, which may reduce the survival of incubating eggs in the gravel and change benthic invertebrate production. Landform and land use both influence mass wasting rates. Most sediment from mass wasting originates from inner gorge landforms (steep, stream-adjacent slopes). On average, these areas cover less than 20% of the mountain basins in the Skagit but produce about 75% of the sediment delivered to streams. Hillslopes >30° are also generally unstable tending to produce shallow-rapid landslides from bedrock hollows or channel headwalls. Hillslopes <30° are generally stable. Deep-seated failures, usually located in glacial deposits or phyllite, have high mass wasting and delivery rates to streams. Compared to mature forest, the increase in mass wasting rates for clear-cut forests and forest road areas averages about 6 and 44 times higher, respectively.	<p><u>Estimating impairment of sediment supply</u>: Average sediment supply for each subbasin estimated based on average sediment supply rates for 13 combinations of geology and vegetation cover (Landsat '93), which were derived from nine sediment budgets conducted within the basin. Using GIS we calculated average current sediment supply for each subbasin, and the average increase over the natural sediment supply for each subbasin (current/natural). Sediment supply process is considered “functioning” where average sediment supply is <100 m³/km²/yr, or where average sediment supply is >100 m³/km²/yr but <1.5 times the natural rate. Sediment supply is “impaired” where average sediment supply is >100 m³/km²/yr and >1.5 times the natural rate.</p> <p><u>Forest road inventory—identify sediment reduction projects</u>: The inventory rates factors that influence road-related landslides and the consequences of landslides. All ratings concerning the likelihood of landsliding are summed, and then multiplied by a rating of the likelihood that significant stream resources will be impacted. The final value, called the risk rating, ranks roads with respect the threat that they pose to salmon habitat. Higher risk ratings indicate greater chance that a road will fail and impact salmon habitat. Final ratings were grouped into three categories of risk. A rating >30 is high, 16 to 30 is moderate, and ≤15 is low.</p>	Collins et al. 1994, Dietrich et al. 1989, Lisle 1982, Lisle 1989, Lunetta et al. 1997, Madej and Ozaki 1996, Paulson 1997, Peterson et al. 1992, Renison 1998, Sidle et al. 1985

Table C-1 continued. Summary of background and methods for rating individual landscape processes.

Background	Method description	References
<p><i>Riparian Function</i></p> <p>Clearing of riparian forests can alter large woody debris (LWD) recruitment to streams, which in turn alters the habitat characteristics of streams. Reduced LWD recruitment persists for several decades, leading to declining LWD abundance in the first few decades and sustained low LWD abundance between 50 and 100 years after the disturbance. A change in LWD abundance alters fish habitat characteristics such as pool spacing, pool area, and pool depth, and this alteration of habitat characteristics causes changes in the salmonid carrying capacity of a stream.</p>	<p><u>Remote sensing assessment</u>: Riparian forests that can produce 80% of potential late-seral LWD recruitment over time (> 40 m wide) are considered “functioning.” Riparian forests producing 50% to 80% of the potential late-seral recruitment (20 to 40 m wide), are considered “moderately impaired.” Buffer widths less than 20 m are considered “impaired.” We estimated the proportion of impaired, moderately impaired, and functioning riparian forests by using LANDSAT classifications of vegetation.</p> <p><u>Field inventory</u>: Ratings are the same as above. In addition to documenting forested buffer width, field inventories also classify stand types by species mix and seral stage, which provides sufficient information to prescribe generalized management regimes for each segment of riparian forest. Inventories also identify areas of livestock access and potential fencing projects.</p>	<p>Abbe and Montgomery 1996, Beechie and Sibley 1997, Bilby and Ward 1991, Hicks et al. 1991, Lunetta et al. 1997, Montgomery et al. 1995, Murphy and Koski 1989, Murphy et al. 1985</p>
<p><i>Channel-floodplain</i></p> <p>Disconnecting rivers from their floodplains changes the ability of rivers to supply, transport, or store one or more of the inputs: water, sediment, and wood. This constrains the formation and maintenance of habitat within floodplains. Streambank hardening (hydromodification) prevents channel migration, reduces LWD recruitment, and typically narrows and steepens channels, increasing both sediment and water transport rates. Mainstem channels in the Skagit dominated by hydromodification exhibited less diversity in edge habitat types and less edge habitat area than non-hydromodified mainstem reaches. Juvenile chinook and coho salmon abundance was strongly correlated to wood and other natural cover types when compared to riprap or rubble cover, commonly used for streambank hardening.</p>	<p>Floodplain areas were delineated where the 100-year floodplain was greater than two channel widths using Federal Emergency Management Agency maps or U. S. Geological Survey 7.5-minute quadrangles and aerial photographs. Reach breaks were based on differences in floodplain width and changes in channel pattern. Hydromodified areas were delineated on copies of aerial photos by rafting or jet boating each main channel within floodplain reaches, then digitized into a GIS arc theme.</p>	<p>Beamer and Henderson 1998, Hayman et al. 1996, Ward and Stanford 1995</p>

Table C-1 continued. Summary of background and methods for rating individual landscape processes.

Background	Method description	References
<p><i>Isolated habitat</i></p> <p>Isolation of habitat by levees and culverts has dramatically reduced carrying capacity of the Skagit Basin over the past 150 years. This includes blockages that impede upstream migration of adult salmon seeking suitable spawning areas as well as blockages to other life stages such as juvenile rearing habitat in both the freshwater and estuarine environment. Some isolated habitat can be recovered by simply removing the barrier (e.g., re-building road crossings that block passage), whereas others will require feasibility studies to determine a range of possible alternatives to accommodate both fish use and existing land use.</p>	<p>Man-made barriers to anadromous fish habitat are identified through a systematic field inventory of channel crossing structures (culverts, tidegates, bridges, dams, and other man-made structures). The inventory identifies the type and physical dimensions of structures as well as physical attributes necessary for modeling water flow conditions and comparing results to passage criteria for salmonids.</p>	<p>Beechie et al. 1994, Collins 1998, WDFW 1998</p>

Table C-2. Designation of generalized habitat types. Key habitat is critical (i.e., required for the persistence of a dominant life history type) for at least one life stage or is preferred habitat type by the majority of life stages considered. Secondary habitat (sec) does not provide critical habitat for any life stage and is not a preferred type by the majority of life stages considered.

Reach-level habitat type	Chum	Coho	Chinook	Steel-head	Pink	Total number of life stages examined	Percent of all life stages designated key or critical	Overall designation for pristine habitat
<i>Tributaries reaches:</i>								
Ponds (including beaver ponds and other wetlands with significant open water area)	sec	critical	key	key	sec	10	60%	key
Pool riffle	key	key	key	key	key	10	90%	key
Forced pool riffle	sec	key	key	key	key	10	85%	key
Plane bed	sec	sec	sec	sec	sec	10	0%	sec
Step-pool	sec	sec	sec	key	sec	10	15%	sec
Cascade	sec	sec	sec	key	sec	10	15%	sec
<i>Main river reaches:</i>								
Main channel floodplain < 2 channel widths	sec	sec	sec	key	sec	10	15%	sec
Main channel floodplain > 2 channel widths	key	sec	key	key	key	10	80%	key
Off-channel habitat (e.g., ponds, sloughs, side channels, oxbow lakes)	key	critical	key	sec	sec	10	60%	key
<i>Estuary:</i>								
Estuarine or tidally influenced wetlands	key	sec	critical	sec	sec	5	40%	key
Blind channel	key	key	critical	sec	sec	5	60%	key
Subsidiary channel	key	key	key	sec	key	5	80%	key
Main channel	key	key	key	sec	key	5	80%	key

Table C-3. Magnitude of peak flows for the lower Skagit River before and after flood storage capability.

Flood return period (years)	Before flood storage (cms)^a	After flood storage (cms)^b
2	3,147	1,830
5	4,735	2,479
10	5,862	2,934
25	7,361	3,540
50	8,528	4,015
100	9,734	4,508

^a Skagit River near Sedro Woolley (river km 36), reported in Williams et al. 1985.

^b Skagit River near Mt. Vernon (river km 25), Sumioka et al. (1998).

Table C-4. Distribution of 234 field-sampled riparian sites by GIS-based land cover type.

Buffer width class	Late-seral forest (n=24)	Mid-seral forest (n=13)	Early-seral forest (n=24)	Other forest (n=96)	Non-forest (n=77)
< 20 m forested buffer “impaired”	0%	8%	8%	42%	90%
20-40 m forested buffer “moderately impaired”	0%	0%	4%	15%	6%
> 40 m forested buffer “functioning”	100%	92%	88%	43%	4%

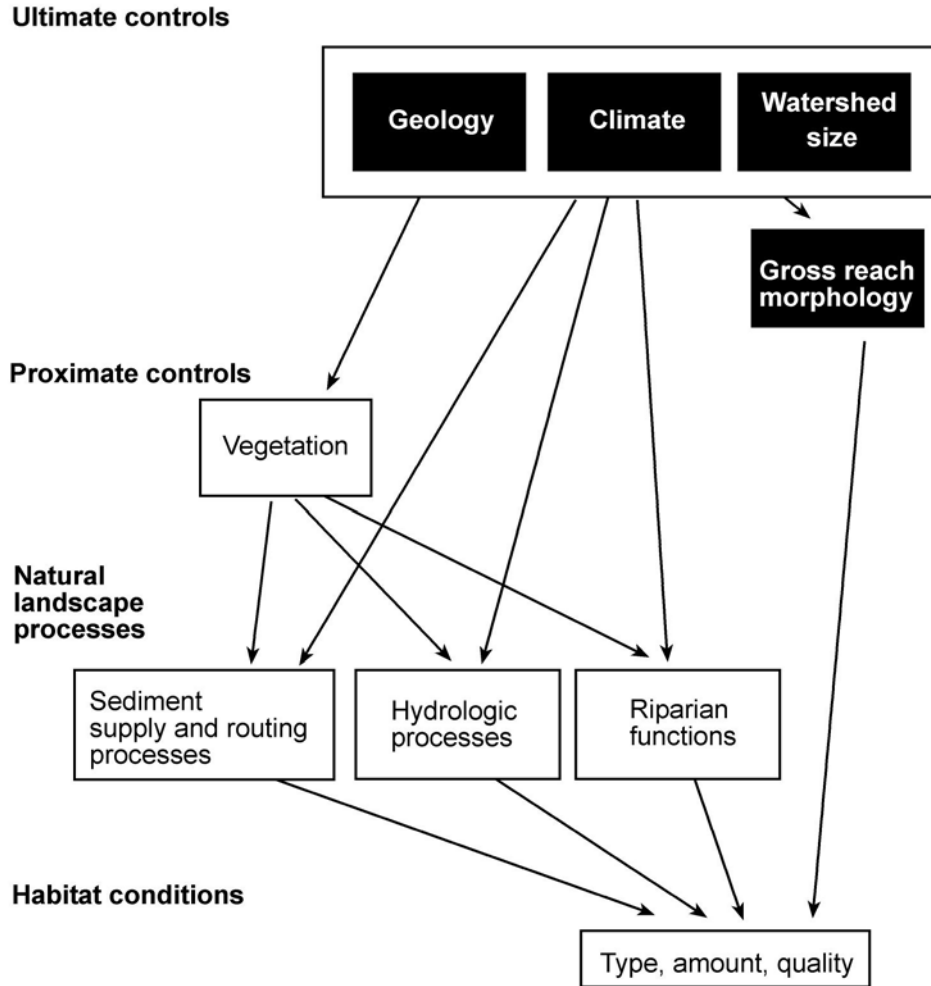


Figure C-1. Simplified flow chart depicting interactions between watershed controls and processes resulting in physical habitat conditions. Shaded boxes represent components that are not influenced by land and resource management while unshaded boxes represent components that are influenced by land and resource management.

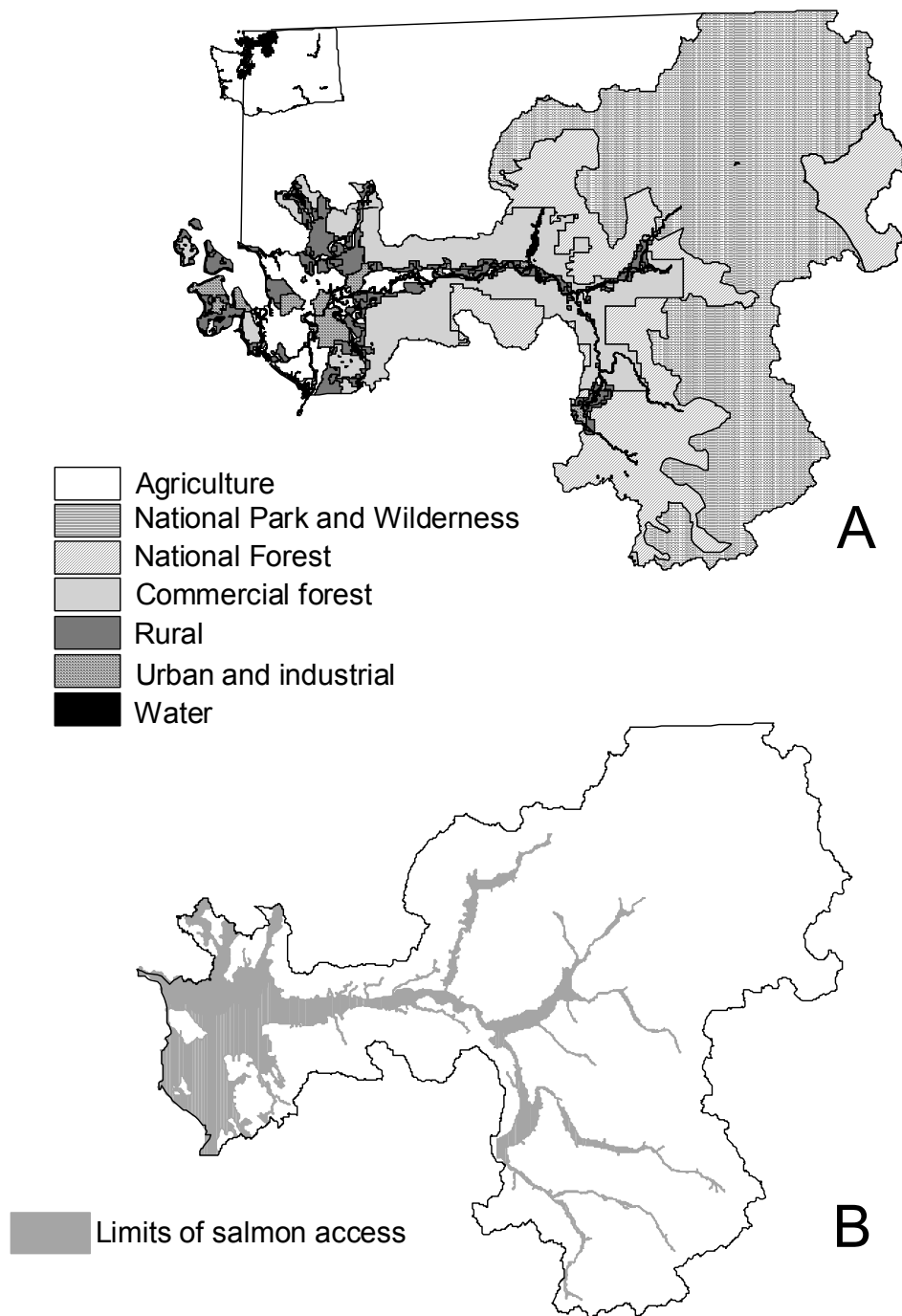


Figure C-2. Location, land use pattern (A), and area of historical salmon access in the Skagit River Basin in Washington State (B).

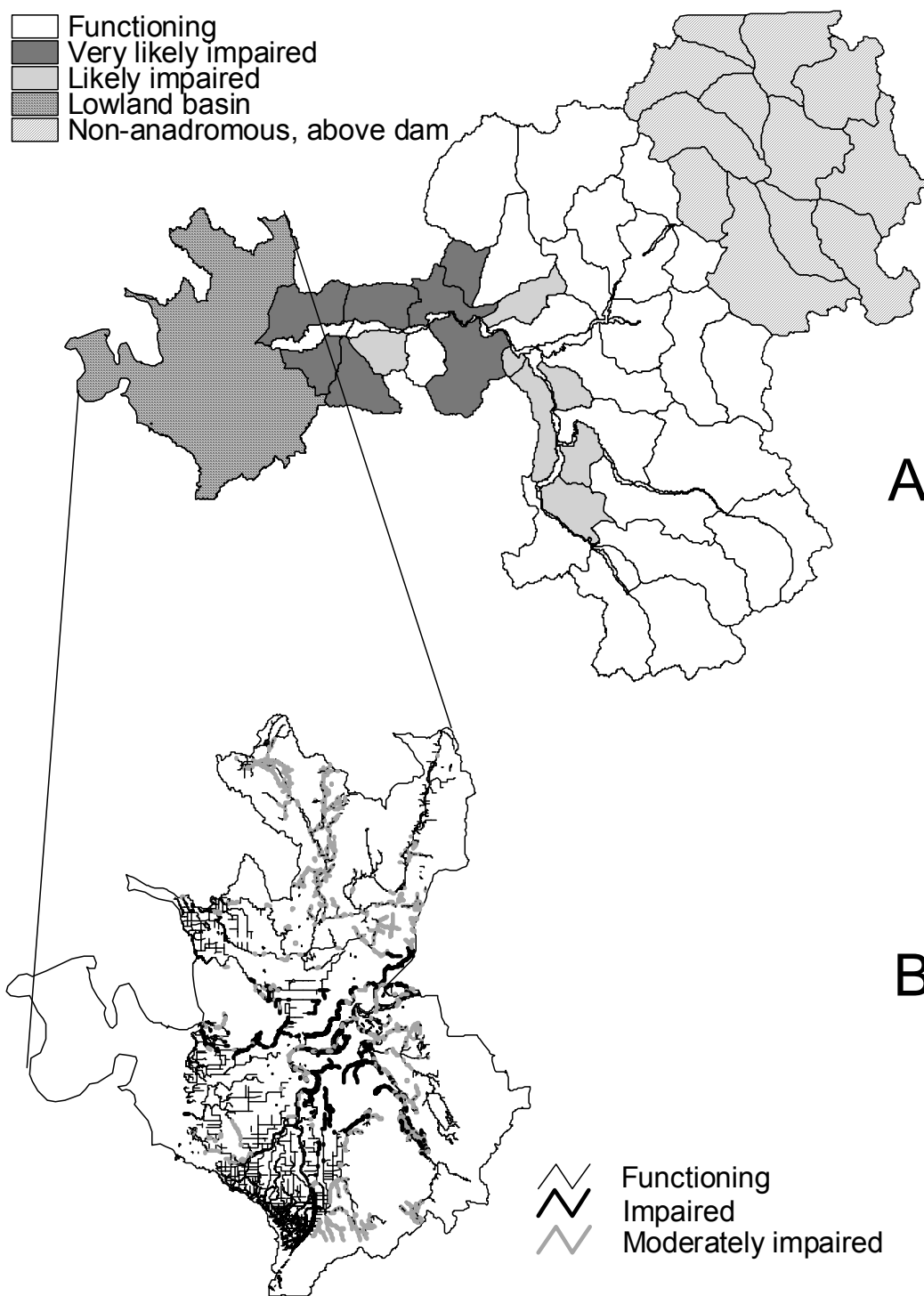


Figure C-3. Subbasins in forested mountain areas of the Skagit River Basin where peak flow is likely impaired (A), and streams in lowland basins where peak flow is planned to be impaired (B).

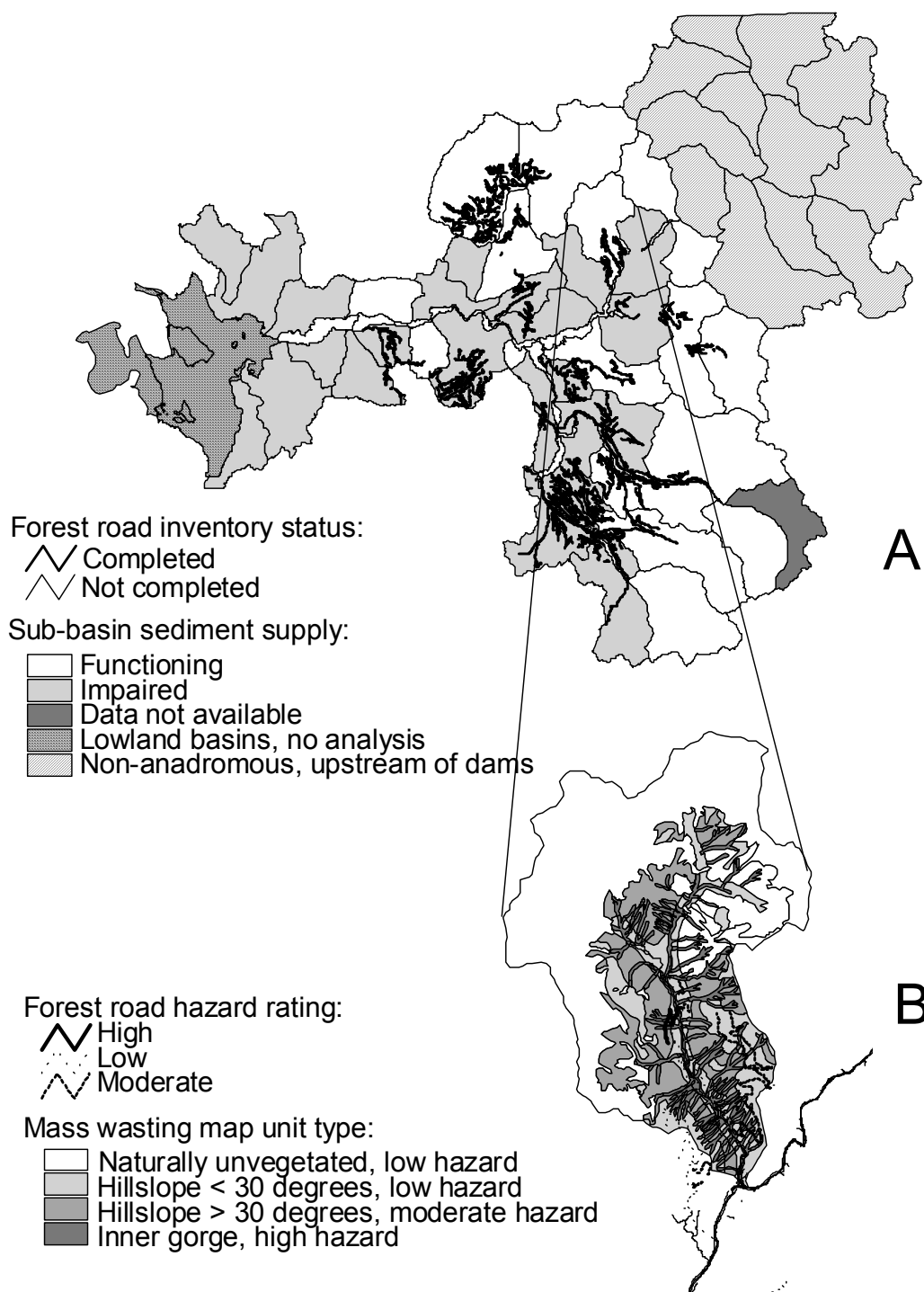


Figure C-4. Sediment supply ratings for mountain basins and status of forest road inventory throughout the Skagit Basin (A), and example of detail for road segments and landslide hazard units in the Bacon Creek watershed (B).

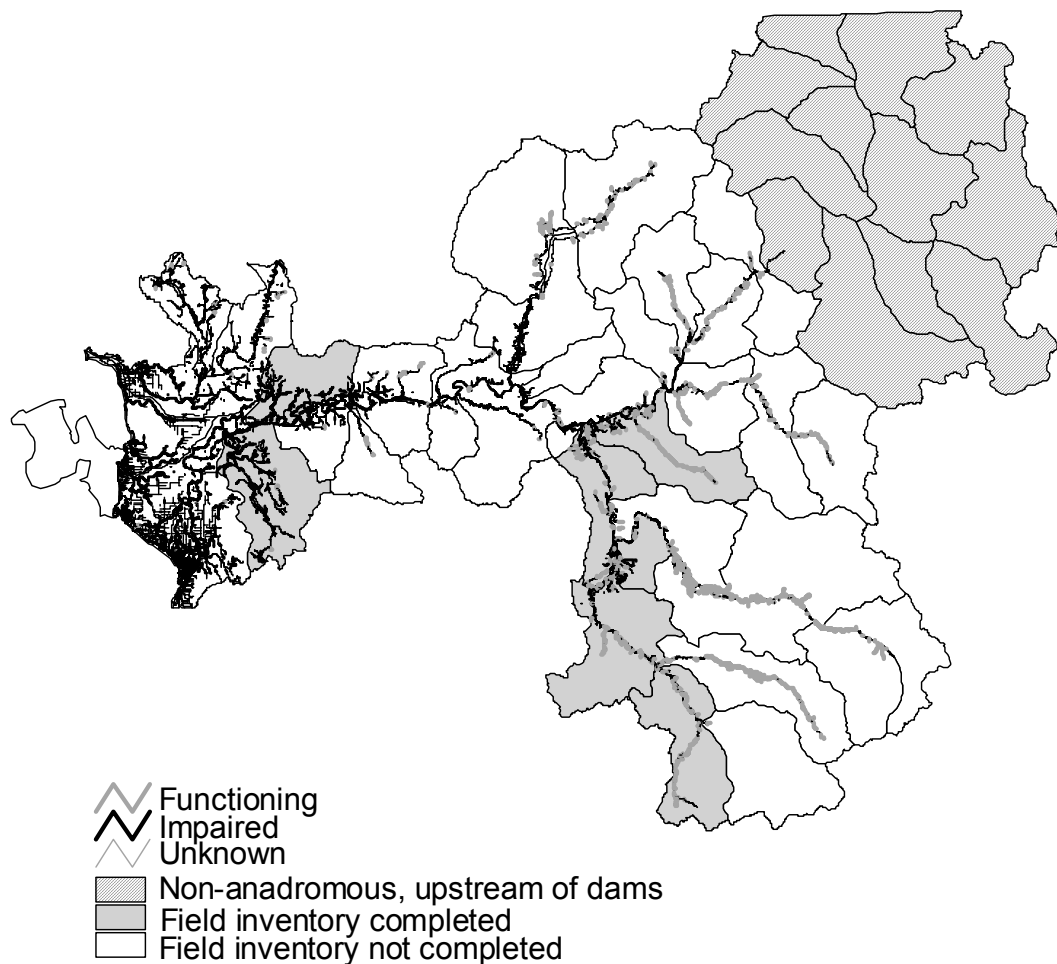


Figure C-5. Map of riparian areas likely impaired or functioning in the Skagit River Basin. Shaded subbasins are where field-based riparian inventories have been completed.

Figure C-5. Map of riparian areas likely impaired or functioning in the Skagit River Basin. Shaded subbasins are where field-based riparian inventories have been completed.

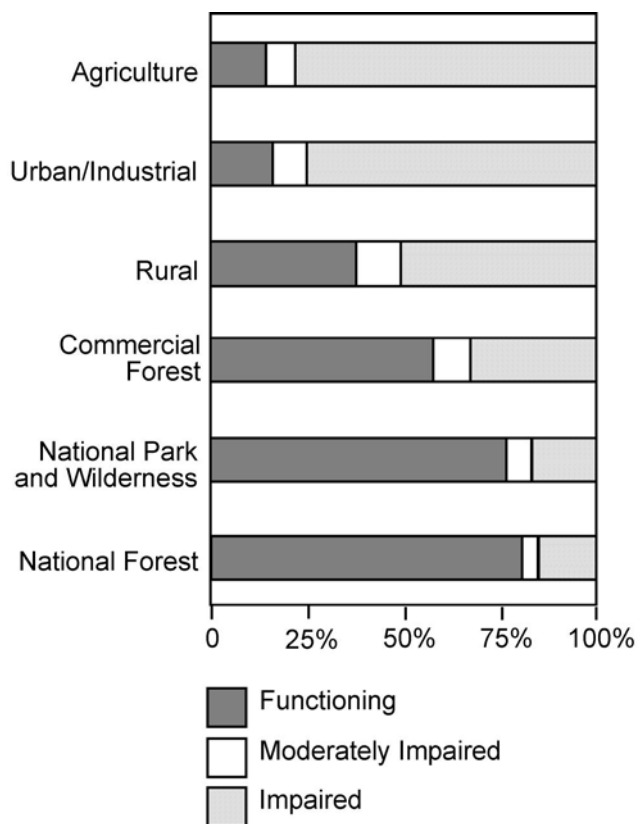


Figure C-6. Estimated percentage of riparian category (impaired, moderately impaired, and functioning) along non-mainstem channels in the anadromous zone of the Skagit River Basin.

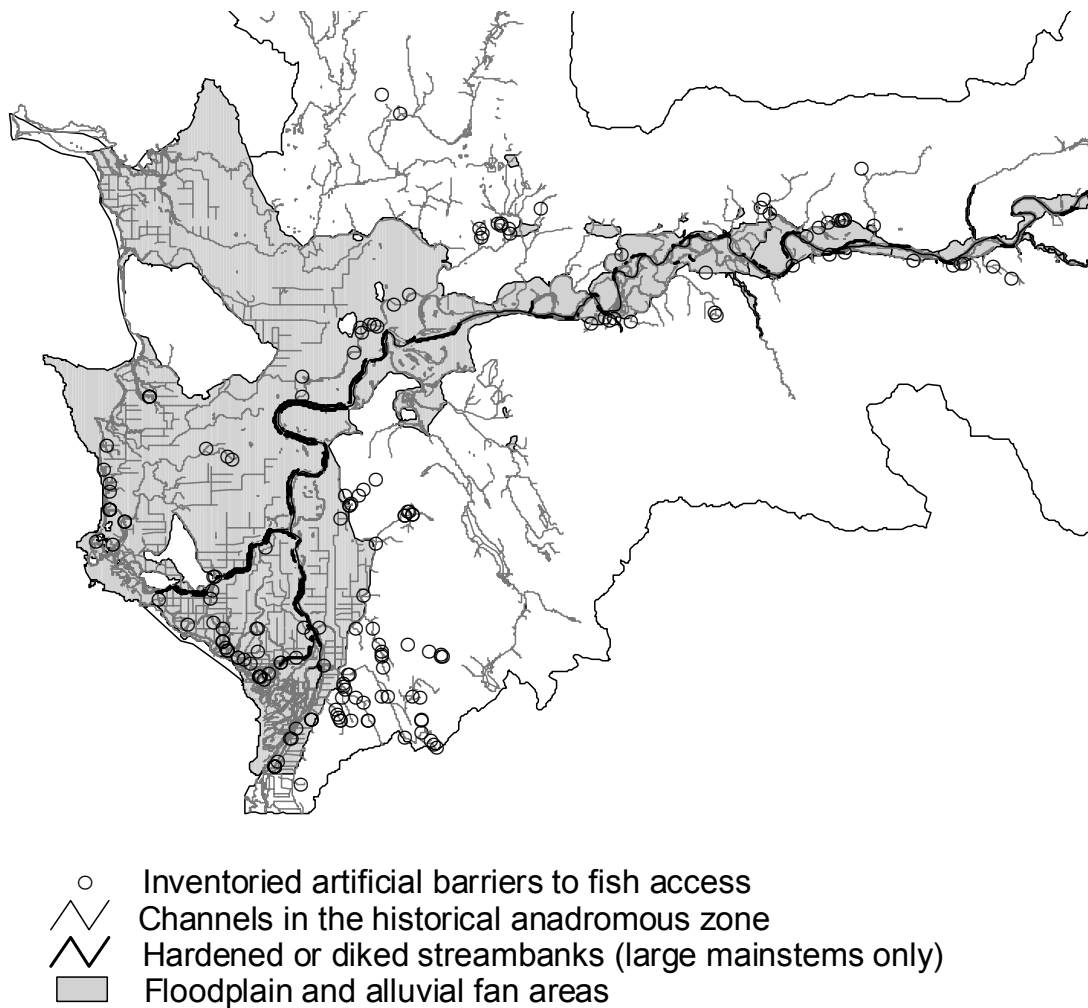


Figure C-7. Location of hydromodification and man-made barriers (Lower Skagit Basin only).

Figure C-7. Location of hydromodification and man-made barriers (Lower Skagit Basin only).

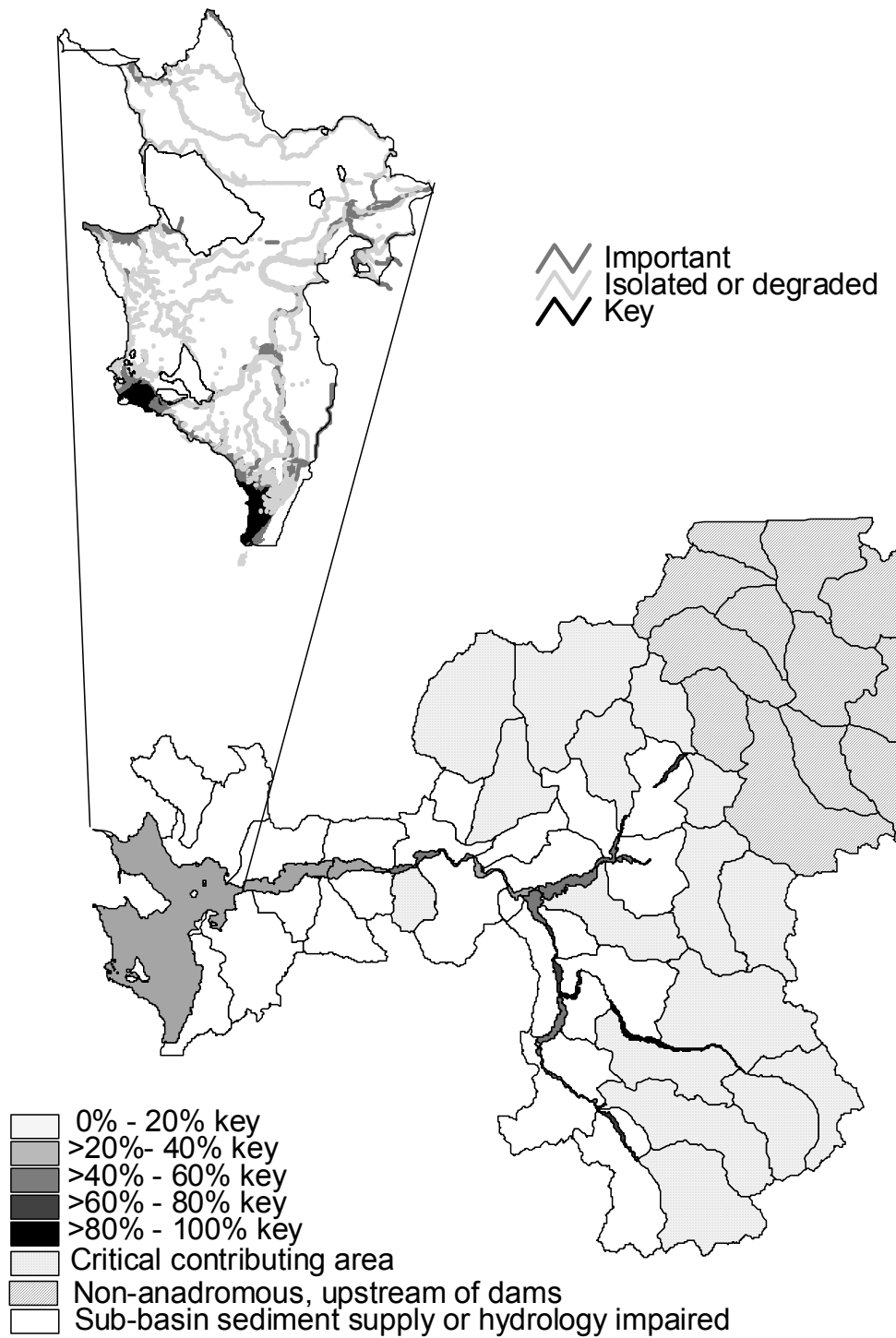


Figure C-8. Distribution of generalized habitat types throughout the Skagit River Basin with example of detail for the delta region.

*** END OF REVIEW DRAFT ***